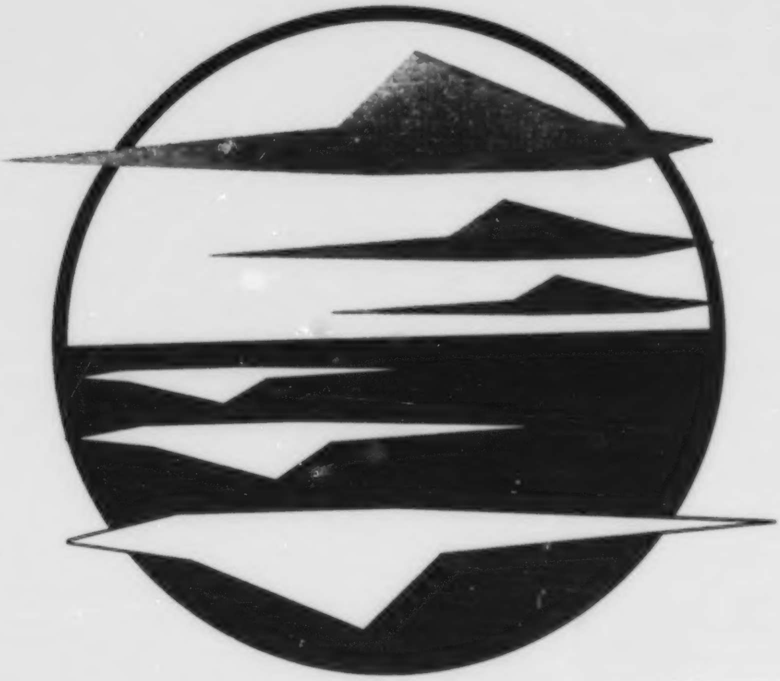

Monitoring Status and Trends In Marine Environmental Quality

Proceedings of a symposium in conjunction with
the 17th Annual Aquatic Toxicity Workshop,
Vancouver, British Columbia

November 5 – 8, 1990

Edited by
Lee E. Harding

Prepared for Environment Canada
by Richard Banner
Polestar Communications

IN MEMORY OF

MICHAEL WALDICHUK

Dr. Michael Waldichuk suddenly and tragically passed away on May 4th, 1991. He was one of the world's pre-eminent marine environmental scientists. Born in Bukovina, Romania, in 1923, he received his B.A. and M.A. in Chemistry from the University of British Columbia and earned the University of Washington's first Ph.D. in Oceanography in 1955. He worked for the Fisheries Research Board of Canada at Nanaimo and West Vancouver, then for the current Department of Fisheries and Oceans as a senior scientist and as the first Director of the West Vancouver Laboratory. Throughout the 1960's, 1970's and 1980's he published many primary and review papers on marine pollution. He served as a founding scientist and long-term member of the United Nations Joint Group of Experts on Scientific Aspects of Marine Pollution (GESAMP), and was its chairman from 1971 to 1973. He most recently contributed to the 1990 State of the Marine Environment report of GESAMP, reporting on the marine environmental quality of the Northeast Pacific.

Mike was a good friend and mentor to many of us at Environment Canada and will be sorely missed. His chairmanship of the platform session and workshop on marine environmental quality monitoring, reported here, will contribute to his lasting legacy for the protection of the marine environment.

DSS Cat. no. EN40-11/025-1990E

ISBN 0-662-18994-9

92-00343/

ABSTRACT

A Marine Environmental Quality (MEQ) Monitoring Symposium was held in November, 1990, in conjunction with the 17th Annual Aquatic Toxicity Workshop in Vancouver, B.C. Following formal presentations of invited papers on key elements of an MEQ status and trends program, participants analyzed selected case histories in a workshop format. At a separate meeting on November 7, 1990, an expert panel evaluated a proposed Canadian "Mussel Watch" program. The MEQ Advisory Group Monitoring Task Group of Environment Canada met on November 8, 1990, to further develop the proposed program within a broader jurisdictional framework. Scientists and staff of DFO participated in the workshop and meeting, in addition to the DOE Task Group members; as well representatives of the United States National Oceanic and Atmospheric Administration and the British Columbia Ministry of the Environment attended to provide international and provincial perspectives

The Symposium was designed to have leading experts present key papers on specific aspects of marine environmental quality (MEQ) monitoring relevant to a proposed Canadian MEQ monitoring program. They included talks on pollution sources, environmental effects monitoring design strategies, direct measures and derived indices, MEQ objectives, sampling and analysis and data management considerations.

This volume contains (a) full manuscripts, extended summaries or abstracts of selected papers given at the platform session on Marine Environmental Quality Monitoring; (b) a summary of the workshop discussions, (c) a summary of the Mussel Watch Expert Panel discussions and (d) notes of the November 8 Monitoring Task Force meeting.

RÉSUMÉ

Un symposium sur la surveillance de la qualité de l'environnement marin a eu lieu en novembre 1990 à Vancouver, en Colombie-Britannique, conjointement avec le 17^e colloque annuel sur la toxicité aquatique. Après la présentation formelle des exposés des conférenciers portant sur les éléments clés d'un programme sur l'état et les tendances en matière de qualité de l'environnement marin, les participants ont analysé en atelier certains cas passés. Lors d'une autre réunion tenue le 7 novembre 1990, un groupe d'experts a évalué un programme canadien de surveillance des moules qui a été proposé. Le groupe de travail sur le surveillance du Groupe consultatif sur la qualité de l'environnement marin d'Environnement Canada s'est réuni le 8 novembre 1990 pour élaborer davantage le programme proposés de façon à ce qu'il s'inscrive dans un cadre juridictionnel plus large. Des scientifiques et d'autres fonctionnaires du MPO ont participé au colloque et à la réunion, en plus des membres du groupe de travail d'EC; des représentants de la United States National Oceanic and Atmospheric Administration et du ministère de l'environnement de la Colombie-Britannique y ont également été invités pour les perspectives internationales et provinciales qu'ils étaient en mesure de livrer.

Le symposium visait à ce que des experts reconnus viennent y traiter de questions fondamentales touchant la surveillance de la qualité de l'environnement marin pour faire avancer le dossier du programme canadien en cette matière actuellement proposé. Il y a eu des discussions sur les sources de pollution, les stratégies concernant la conception de la surveillance des effets sur l'environnement, les mesures directs et les indices dérivés, les objectifs de la surveillance de la qualité de l'environnement marin, l'échantillonnage et les analyses et la gestion des données.

Le présent volume contient: (a) le texte complet, un sommaire ou un résumé détaillés de certains des exposés formels présentés lors de la premièrs partie du colloque sur la surveillance de la qualité de l'environnement marin, (b) un sommaire des discussions en atelier, (c) un sommaire des discussions du groupe d'experts sur la surveillance des moules et (d) des notes sur la réunion du group de travail tenue le 8 novembre.

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PREFACE

Conservation and Protection Service of Environment Canada approved, in October, 1987, a Framework for Marine Environmental Quality Management (MEQ). In approving this Framework, the Management Board instructed the MEQ Advisory Group (MEQAG) to prepare an Action Plan. This Action Plan identified a priority to develop an MEQ status and trends program for Canada.

A preliminary status and trends program addressing the effectiveness of C&P programs to limit entry of pollutants into the marine environment was drafted in 1988. However, Environment Canada and Fisheries and Oceans Canada subsequently agreed to develop the program jointly, from a broader jurisdictional perspective.

An MEQ Monitoring Workshop was held in November, 1990, in conjunction with the 17th Annual Aquatic Toxicity Workshop in Vancouver, B.C. At a separate meeting, a panel of experts reviewed the objective and design of a proposed "mussel watch" programme for Canada, prepared previously under contract. Following formal presentations of invited papers on key elements of an MEQ status and trends program, participants analyzed selected case histories in a workshop format. Following this, the MEQAG Monitoring Task Group met on November 8, 1990, to further develop the proposed program within this broader jurisdictional framework. Scientists and staff of DFO participated in the workshop and meeting, in addition to the DOE Task Group members; as well representatives of the United States National Oceanic and Atmospheric Administration and the British Columbia Ministry of the Environment attended to provide international and provincial perspectives.

This volume contains (a) full manuscripts, extended summaries or abstracts of selected papers given at the platform session on Marine Environmental Quality Monitoring; (b) a summary of the workshop discussions, and (c) summary of the mussel watch expert panel discussion and (d) notes of the November 8 meeting. The summary and notes were prepared by the rapporteur, Richard Banner of Polestar Communications.

SYNTHESIS: PRESENTED PAPERS ON MEQ MONITORING

INTRODUCTION

The Symposium was designed to have leading experts present key papers on specific aspects of marine environmental quality (MEQ) monitoring relevant to a proposed Canadian MEQ monitoring program. These key aspects were areas of scientific uncertainty or controversy, or where program design could benefit from formal presentation and discussion of alternative methods or design strategies. They included talks on pollution sources, environmental effects monitoring design strategies, direct measures and derived indices, MEQ objectives, sampling and analysis and data management considerations.

The talks highlighted in this volume were selected from among all the papers given at the Annual Aquatic Toxicity Workshop on the basis of their relevance to MEQ monitoring. They include all the papers given in the MEQ monitoring session chaired by Mike Waldichuk and Martin Pomeroy, as well as selected papers from other sections. Full proceedings of the Aquatic Toxicity Workshop will be published separately.

POLLUTION SOURCES

Provinces, Territories and federal authorities maintain information on permitted waste discharges. Much of this information derives from permits issued by provincial or territorial governments, or other permitting agencies (such as Indian and Northern Affairs Canada for Arctic offshore drilling, Energy, Mines and Resources for west and east coast offshore drilling, etc.). Environment Canada issues permits for ocean dumping and maintains a database of materials approved for dumping, including contaminant levels in dredged material. These varied data sources form the first line of information on threats to marine environmental quality, on which pollution fate and effects monitoring programs can be based.

Merely identifying the sources, however, does not fully describe the threats to marine environmental quality. At the Symposium, Anderson et al described an effluent and ambient toxicity assessment for San Francisco Bay that would seem to have application in Canadian industrial harbours, such as Halifax Harbour and Vancouver Harbour. Data from such a process is needed to link the discharges with observed effects.

GESAMP (1987) showed that riverine input of pollutants is significant globally, and may be very important regionally. Relatively little of the pollution load reaches the open ocean, most being retained in estuaries and recycled through coastal ecosystems by such processes as adsorption-desorption and precipitation-dissolution reactions, biological uptake and microbial degradation.

From past work (MEQAG, 1988) it is clear that **large improvements in riverine contaminant monitoring are needed to quantify inputs to the Canadian marine environment.** While reliable flow data are available from many rivers in Canada, only a few are monitored for contaminant levels from which loadings to the ocean might be calculated. MEQAG (1988) had previously proposed the Exploits, St. Lawrence, Churchill, Mackenzie and Fraser Rivers as important riverine sources to include in a proposed MEQ monitoring program. Other candidates include the Saint John, Saguenay and Restigouche Rivers.

At the Symposium, Byrd gave an analysis of sampling and analytical constraints to determining riverine flux of contaminants to the marine environment, based on a study of large and small rivers along the east coast of Canada and the U.S. He showed that storm related events have a large impact upon the transport of material by rivers and may account for the majority of the annual riverine flux of a substance. The concentrations of some constituents vary with discharge, some reflecting dilution by the larger flow, and some increasing through the storm period, reflecting a flushing effect of material stored in the drainage basin. Depending upon intervals between storm events, antecedent effects from previous storm events may reduce the amount of transport in successive storm events. **Riverine sampling to support MEQ monitoring must take these transient events into account.** As with other monitoring programs, data quality assurance is critical.

EFFECTS MONITORING DESIGN STRATEGIES

A number of papers were given on how MEQ monitoring strategies and design can be related to program goals. From these talks, several themes emerged: First, as previously discussed by Phillips and Segar (1986) and others, **spacial and temporal scales need to be resolved**, and these relate to program goals as well as to rate/scale of change in environmental variables: It is no good setting up a once-annual program for parameters such as riverine contaminant flux or phytoplankton blooms that pass through complete cycles in days or weeks; nor can multi-annual events, such as a 50 year flood or unusual drought, be ignored in site-specific monitoring. At one end of the scale, Nelson showed how "real time" data (16 hour data turn-around) can be used for management decisions while sensitive dredging projects are occurring. Taylor also discussed real-time monitoring to warn aquaculturists of phytoplankton blooms that have killed millions of dollars worth of farmed salmon in B.C. With even a few hours warning, salmon farmers can take a number of defensive actions to avoid or limit losses.

At the other end of the spacial and temporal scales, Paul et al outlined EPA's plan to monitor status and trends in ecosystems throughout the U.S. coastal waters on a regional scale over decades. EPA will monitor **indicators** of the condition of ecological resources, pollutant exposure and habitat condition and seek associations between man-induced stresses and ecological conditions. Adams et al showed how specific **measures of ecosystem health** (see below) can be incorporated into large scale monitoring program design of the NOAA State and Trends "mussel watch" (contaminants in shellfish) program. Mearns also gave a perspective on how the NOAA Status and Trends program has fared: Long term inconsistency of methods and poor availability of data and supporting information have inhibited successful assessment of long term trends of U.S. coastal waters. **Once a monitoring program begins, all effort should be made to continue it** and to resist change even in the face of changing technology. New techniques should supplement, rather than replace, the primary monitoring procedures.

These talks on NOAA's Mussel Watch were excellent background at a separate meeting on November 7. A Panel of Experts formed by Environment Canada and the Department of Fisheries and Oceans met to consider implementation of a proposed Canadian mussel watch program, prepared under contract to Environment Canada (Sojo et al., 1990) (Appendix 2). The purposes were to (a) discuss Environment Canada's and the Department of Fisheries and Oceans' objectives in monitoring contaminants in shellfish, and (b) evaluate program design, specifically parameters to analyze and sampling and analytical protocols. **It was agreed to proceed with monitoring of contaminants in commercial shellfish species on a trial basis in commercial harvest areas, using the NOAA protocols** (Appendix 2). Results will complement other monitoring of contaminants in seabird eggs by the Canadian Wildlife Service and of contaminant levels in marine mammal tissues by the Department of Fisheries and Oceans.

The second major theme, as Whitfield and Valiela emphasized, is that **monitoring for sustainable development needs to measure system attributes such as resilience, trophic structure, productivity and energy flow to determine ecosystem state; and monitoring should not only assess ecosystem state, but also identify processes that lead to state changes.** Expanding on this theme, Sadler and LeBlanc said that a greater emphasis should be given to regional thresholds and the cumulative relationship between total stresses and ecosystem integrity. Schaeffer pointed out how ecosystem health differs from health of an individual, and gave examples of a systematic framework to diagnose ecosystem "disease" by detecting exceedence of ecosystem threshold criteria.

Finally, no agency or network of agencies can monitor all components of marine ecosystems. Marine ecosystems and threats to them are sufficiently diverse that indices appropriate in one situation may not work in another. Program design therefore needs to include a variety of indices of MEQ appropriate to the region and to national/regional program goals.

MEASURES OF MARINE ENVIRONMENTAL QUALITY

Papers on measures – indicators which are directly measurable, rather than indices which are derived – of MEQ included papers on exposure and effects at the cellular, organism and population levels. GESAMP (1990) reviewed literature from around the world and concluded that long term biological changes can be attributed to contaminant exposure at low level, or as a result of slow buildup of contamination in the environment or in target species. The use of biota to monitor contaminant exposure and bioaccumulation was discussed by Salazar and Salazar (mussels/tributyltin), Adams et al (direct and indirect mechanisms of contaminant stress on fish populations), Tay et al (bioaccumulation in *Macoma*), Elliott et al (contaminants in seabirds), Lauenstein (contaminants in bivalve molluscs), Muir and Norstrom (PCBs and dioxins/furans in marine mammals), Mearns (NOAA Status and Trends Program) and Pocklington (contaminants in polychaetes). Myers et al (liver lesions in flatfish), Goyette (liver lesions in English sole), Adams et al (measures of chronic stress in fish), Tay et al (liver lesions in flatfish), Collier et al (MFO in benthic fish), Addison (MFO), and Imber et al (subcellular indices of MEQ) gave papers on metabolic and histological indicators of ecosystem stress. Ciammaichella et al (*Neanthes* growth) and Pocklington (polychaete community structure) spoke about infauna production of biomass and community structure as pollution effect indices.

The techniques outlined by these authors were among the biological variables reviewed by GESAMP (1980) in the context of strategies for marine pollution monitoring. They are all potential indicators of pollution stress in fish; however, very little hard information is available on how these effects are translated into effects on populations, and still less on ecosystem responses. Perhaps their best utility is in early warning systems, as outlined by Cairns and Van der Schalie (1980). Adams and his co-authors showed that this approach is valid, because effects at the cellular and tissue levels are promulgated upward through increasing levels of biological organization, manifested as changes in growth and reproduction.

From this myriad of effects measures one can draw several conclusions. First, we need a suite of tests that include **exposure** and **effects** of contaminants. The best measures, from an ecosystem health point of view, might be tissue analyses in the higher trophic levels (seabirds and marine mammals), coupled with effects indices such as reproductive performance.

Second, an MEQ Status and Trends network must measure **processes**, such as bioaccumulation, biomass production, respiration (as affected by biochemical oxygen demand, and/or its resulting hypoxia), reproduction, mortality and eutrophication that, if impaired, may lead to changes in ecosystem state. **Significant changes should be detected early so as to influence management decisions.**

Lastly, it must include effects that promulgate upward to affect the **structure** and key **system attributes**, such as resilience, of marine ecosystems. How these measures can be incorporated into a marine environmental monitoring program that will serve environmental decision-making objectives and provide information to describe the state of the environment, is a formidable task.

INDICES OF MARINE ENVIRONMENTAL QUALITY

O'Connor and Dewling (1986) provide a good analysis of the use of indices of marine degradation versus direct measures. Indices, derived from direct measures, are readily interpretable not only by the scientist, but also by managers and laypersons. They can combine impacts of several pollutant sources, providing a consistent basis for management decisions.

Indices of marine environmental quality were discussed by Davis and Gaudet, McRae and Mearns. Davis and Gaudet described uses of marine environmental quality indicators in environmental decision-making, including environmental impact assessment, site rehabilitation, state of the environment reporting and the development of standards for aquatic ecosystem protection. Environmental quality indicators will facilitate the transition from traditional "end of pipe" regulatory approaches based on best available or practicable technology, to those designed to sustain environmental quality. McRae outlined national efforts to decide upon a set of nationally relevant indicators of marine environmental quality. Mearns showed how the U.S. Status and Trends Program has used direct measures as MEQ indices: contaminant levels in mussels were compared for all regions of the country, and "hot spots" were identified. **It was clear from these papers and the ensuing discussions that more work is needed to develop and standardize indicators of marine environmental quality that will be relevant in both an environmental management and a state of the environment reporting context.**

MARINE ENVIRONMENTAL QUALITY OBJECTIVES

The talks by Gillam, Cross et al, Zarba, Word and Ward, Albright and Malek, Dexter, and Long described and evaluated procedures to establish marine sediment quality objectives (most of these were presented and discussed at separate sediment quality guidelines and bioassay workshops and a panel discussion). Procedures varied, but most were based on both chemical and toxicological data while some (Apparent Effects Threshold and Triad approaches) also used benthic community data. Equilibrium partitioning (Zarba) used a toxicokinetic approach to estimate contaminant concentrations in sediment porewater, for comparison with known *in vitro* toxicity. These approaches give remarkably convergent results, considering how fundamentally different they are. Nevertheless, the selection and application of these various approaches still requires considerable judgement. While all of them produced useful results, consistency of marine sediment quality criteria and standards between regions and among jurisdictions is still a long way off. As well, development of site-specific objectives requires a considerable investment in data collection and analysis. Speakers, questioners, panellists and workshop participants agreed that continued development of these approaches, standardization of the bioassay techniques that underlie them, and implementation of QA/QC procedures to ensure consistent quality of results, remain urgent.

SAMPLING AND ANALYSIS AND DATA MANAGEMENT

Without a commitment to long term monitoring, and without consistent sampling strategies or standard sampling and analytical protocols, data for most locations have been too inconsistent to analyze for site specific trends (Kay, 1989), and too incompatible for regional comparisons. Environment Canada has begun the process of standardizing regional monitoring along the Arctic (e.g., Arctic Laboratories Ltd. 1985), Atlantic (Moores and Barchard, 1988; P. Lane and Associates, 1990) and Pacific (Harding, 1985) coasts.

National standardization is being achieved for pulp mill EEM programs through guidelines, including sampling and analytical protocols and QA/QC procedures, which will be legal requirements of the revised Pulp and Paper Regulations under the Fisheries Act. Protocols currently being examined for compatibility include those in use by state and federal agencies involved in Puget Sound MEQ monitoring (Tetra Tech Inc., 1986.). These protocols, when completed, may be used in or adapted for other program areas such as ocean dump site monitoring and mine discharge site monitoring. Thus, compatibility of data between EEM programs will be improved and some degree of international data compatibility is at least possible.

At the symposium, several presenters discussed statistical approaches to data analysis: Clark (a method for estimating diversity of aquatic communities), Uthe and Misra (Statistical constraints of trends analysis), Cretney (value of chemometrics to data quality and interpretation), Cross et al (a multivariate approach to analyzing spacial sediment chemistry, toxicity and community structure data). Mearns discusses large-scale data management and analysis of NOAA Status and Trends contaminants in shellfish data to detect long term trends in the occurrence and persistence of chemical residues. Long used sediment data from this program to rank sites for the relative potential for contaminant-induced effects, using matching toxicological and sediment chemistry data. Aside from the value of discussing and comparing various techniques, the main points deriving from these talks are (1) the need for *a priori* statistical design in any effects or ambient marine environmental monitoring program, and (2) the need to make judgements on the design, given the lack of standardization of statistical approaches.

CASE STUDIES

An excellent analysis of MEQ monitoring was recently completed by the U.S. National Research Council, whose Marine Board analyzed design of three regional MEQ monitoring programs and made recommendations applicable on a national basis (NRC, 1990). Mike Waldichuk, co-chairman with Martin Pomeroy of a special workshop within the MEQ Monitoring Symposium, summarized the main conclusions of this important report . He noted that, as the NRC (1990) report concluded, successful program design depends on the following factors:

- Goals and objectives must be clearly articulated in terms that are meaningful to the public and provide the basis for scientific investigation.

- Attention must be paid, and adequate resources provided over the long term, to the management, synthesis, interpretation and analysis of the data generated by monitoring.

- Quality assurance procedures must include peer review.

- Because even well-designed monitoring results in unanswered questions about environmental processes or human impacts, supportive research must be provided.

- Programs must be sufficiently flexible to allow for modification where changes in conditions or new information indicates the need.

- Monitoring information should be available to all interested parties in a form that is useful to them.

Dr. Waldichuk concluded his introduction by noting NRC's (1990, p. 4) caveat:

> "The committee believes that implementation of its recommendations is vital to better protection, restoration and understanding of the marine environment. Yet it does not wish to overstate the usefulness of monitoring programs. The marine environment is complex and variable and it is often difficult to detect, identify and measure anthropogenic impacts clearly.... When well developed, applied and used, environmental monitoring can help quantify the magnitude of uncertainty, thereby reducing but not eliminating uncertainty in decision making."

In all, 22 papers were presented at the symposium that consisted in part of case histories of marine environmental effects monitoring. Three of these were featured in the separate workshop: Barchard and Johnson Hayden, Mearns and Goyette gave examples of regional MEQ monitoring program designs for the Gulf of Maine, Puget Sound and Vancouver Harbour, respectively. Results of the workshop are summarized in Appendix 1.

One case study, presented in the platform session, deserves special mention. Kay described the monitoring of shellfish growing waters for bacterial contamination, a program that has all the elements necessary in an MEQ status and trends monitoring program. There is a clear objective (protection of human consumers from contaminated products), a commitment to long term sampling, *a priori* sampling and statistical design, rigorous and standardized sampling and analytical protocols, measurement of results against marine water quality standards (faecal coliforms per 100 mℓ of water), clear decision points for management actions based on results (waters exceeding the standard are closed for harvest) and unequivocal indices can be derived (number of hectares closed; length in kilometres of productive shoreline affected; etc.) that make sense in regional and national "state of the environment" contexts. Although this program does not address ecosystem health, it could be used as a design model for programs with broader, ecosystem health objectives.

REFERENCES

Arctic Laboratories Ltd., 1985. *Sampling and analysis in the Arctic marine benthic environment.* Volume 2, Guide to practice. Report for EPS, Yellowknife, N.W.T.

Cairns, J. Jr. and W. H. Van der Schalie, 1980. *Review Paper: Biological Monitoring Part I: Early warning systems.* Water Res. 14: 1179–1196.

Joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP), 1987. *Land/Sea boundary flux of contaminants: Contributions from rivers.* IMCO/FAO/UNSECO/WMO/WHO/IAEA/UN/UNEP Report Series No. 32.

Joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP), 1980. *Monitoring biological variables related to marine pollution.* IMCO/FAO/UNSECO/WMO/WHO/IAEA/UN/UNEP Report Series No. 11.

Joint Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP), 1990. *Long-term consequences of low-level marine contamination: an analytical approach.* IMCO/FAO/UNSECO/WMO/WHO/IAEA/UN/UNEP Report Series No. 40.

Kay, B. 1989. *Pollutants in British Columbia's marine environment.* SOE 89–1.

Harding, L. E. 1986. *West coast marine environmental monitoring program.* Environment Canada unpubl. manuscript.

Marine Environmental Quality Advisory Group (MEQAG), 1988. Appendix 9: Marine Environmental Quality Monitoring: A proposal. In *Marine Environmental Quality Action Plan, Conservation and Protection, 1988 (unpubl).*

Moores, R.B. and W.W. Barchard, 1988. *Point source effects monitoring for the marine environment — bridging the gap between theory and practice.* Seminar, St. John's, Newfoundland, March 15 - 16, 1988. Environment Canada and Memorial University of Newfoundland.

National Research Council (NRC), 1990. *Managing Troubled Waters: The role of marine environmental monitoring.* National Academy Press, Washington, D.C. 125 p.

O'Connor, J.S. and R.T. Dewling, 1986. Indices of marine environmental qualilty. *Environmental Management* 10 (3): 335 - 343.

P. Lane and Associates Ltd., 1990. *Environmental effects monitoring: Pitfalls and possibilities in relation to offshore oil development.* EPS-5-AR-90-1.

Phillips, D. J. H. and D. A. Segar, 1986. *Use of bioindicators in monitoring conservative contaminants: Programme design imperatives.* Mar. Poll. Bull. 17: 10–17.

Sojo, L. E., D. Fyles and M. C. Hamilton, 1990. *National Mussel Watch Program, Phase I: Overview of existing mussel watch programs.* Prepared for Environment Canada by Seakem Oceanography Ltd. Sidney, B.C. 80 pp.

Tetra Tech. Inc., 1986. *Recommended protocols for measuring selected environmental variables in Puget Sound.* Prepared for U.S. EPA, Region X.

Presented Papers on MEQ Monitoring

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Abstract 3

MUSSELS AS BIOINDICATORS: A CASE STUDY OF TRIBUTYLIN EFFECTS IN SAN DIEGO BAY.

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As part of a Navy research program to evaluate the environmental effects of tributyltin (TBT) antifouling coatings, native juvenile mussels were transplanted in San Diego Bay between 1987-1990. Serial seasonal transplants and intensive chemical sampling have documented temporal and spatial variability in TBT and its effects on growth, bioaccumulation and survival that have not been previously reported. The most important aspect of this work, however, is the refinement of mussels as bioindicators and questions that have been raised regarding the environmental significance of the data that may be applicable to all bioindicator responses. A model was also developed that emphasizes the importance of natural factors in modifying both mussel biology and the input of TBT. The mussel bioindicator model illustrates the need to partition those effects to effectively calibrate the bioindicator. The interaction of mussel growth, survival, and bioaccumulation are described in the contest of environmental significance. Overall relationships to laboratory toxicity tests, other biological indicators and environmental monitoring are also discussed.

Abstract 4

AN APPLICATION OF "REAL-TIME" MONITORING IN DECISION-MAKING: THE NEW BEDFORD HARBOR PILOT DREDGING PROJECT.

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A decision-making framework was established for assessing the impacts of a pilot dredging study at the New Bedford Harbor, MA, Superfund site. Concern over possible environmental impacts due to dredging at this site necessitated that a monitoring program be implemented to ensure that unacceptable water quality impacts did not occur during this project. Consequently, criteria were derived, a management committee assembled, and a "real-time" monitoring plan designed. Because many existing chemical concentrations in the water column and indigenous biota exceeded Federal and state water quality limits, site-specific chemical and biological criteria were established. A committee of environmental managers from federal and state government was established with the authority to assess and modify the operation on a daily basis. Finally, a "real-time" monitoring plan was implemented in which water samples were collected, analyzed within 16 hours, and the data supplied to the management committee in order to assess the environmental impact of the previous days' operation. The combined use of site-specific criteria and a "real-time" decision making management process allowed for successful completion of this project with a minimal effect on water quality.

Abstract 5

EPA'S ENVIRONMENTAL MONITORING AND ASSESSMENT PROGRAM: AN ECOLOGICAL STATUS AND TRENDS PROGRAM.

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The U. S. Environmental Protection Agency has initiated the Environmental Monitoring and Assessment Program (EMAP) to monitor status and trends of the nation's near coastal waters, forests, wetlands, agroecosystems, surface waters, deserts and rangelands. The program is also intended to evaluate effectiveness of Agency policies at protecting ecological resources occurring in these systems. Monitoring data collected for all ecosystems will be integrated for national status and trends assessments. The near coastal component of EMAP consists of estuaries, coastal waters, and Great Lakes. Near coastal ecosystems have been regionalized and classified, an integrated sampling strategy has been developed, and quality assurance/quality control procedures and data management designs have been implemented. EPA and NOAA have agreed to coordinate and, to the extent possible, integrate near coastal component of EMAP with NOAA National Status and Trends Program. A demonstration project was jointly conducted in estuaries of the mid-Atlantic states (Chesapeake Bay to Cape Cod) in the summer of 1990. In 1991, monitoring with continue in mid-Atlantic estuaries and will be initiated in estuaries of the Gulf of Mexico.

Abstract 14

EFFLUENT AND AMBIENT TOXICITY PROGRAMS IN THE SAN FRANCISCO BAY REGION.

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The Regional Water Quality Control Board is conducting two study-based programs for evaluating toxicity in discharges and receiving waters of the San Francisco Bay Region. The data from these programs are being used to determine the need for Toxicity Reduction Evaluations, to derive effluent limits, and to refine the Region's Toxicity Control Program.

The ongoing Effluent Toxicity Characterization Program, sponsored by 20 major dischargers, is providing data on species sensitivity, effluent variability and toxicity test precision. The program involves initial screenings of effluents using 11 toxicity tests, followed by repeated testing using batteries of three tests over a one year period. Parallel reference toxicant tests accompany each effluent test.

The Ambient Toxicity program has provided data on the spatial and temporal distribution of toxicity at numerous locations near the margins and in the open waters of the estuary. In some cases, effluent toxicity studies have been linked to ambient toxicity surveys to assess their value in predicting the potential for receiving water toxicity.

Abstract 21

DIRECT AND INDIRECT MECHANISMS OF CHRONIC CONTAMINANT STRESS ON FISH POPULATIONS.

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Chronic contaminant stress can be partitioned into direct and indirect components, both of which can be differentiated by the principal mechanisms of pathways by which they ultimately affect the structure and function of organisms. Direct effects of contaminants are usually expressed first at the cell or tissue level on components such as enzymes, metabolism, osmoregulation, etc. Effects on these lower-level functions are promulgated upward through increasing levels of biological organization and may be ultimately manifested as changes in growth and reproduction. Indirect effects of contaminant stress on organisms are expressed primarily through the food chain via the quality and quantity of energy available to fish. The MFO enzymes and DNA integrity can be used as indicators of direct exposure to contaminants while indicators of nutrition and metabolic energy pools serve as useful indicators of indirect contaminant effects on organisms.

Abstract 24

ENHANCEMENT TO THE PIELOU METHOD FOR ESTIMATING THE DIVERSITY OF AQUATIC COMMUNITIES.

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The Pielou method of estimating the diversity of aquatic communities by plotting the relative abundances of the three most common species per sample on ternary plots have been further developed. Algorithms are presented which permit the ternary plots to be readily produced via any computer spreadsheet have orthogonal graphics capability. Moreover it is shown that the Pielou method may be modified without loss of statistically significant information, so as to present time series diversity data as chronological tables or plots similar to those commonly used for other water quality variables, rather than as the ternary plots used by Pielou. Additionally, the Pielou method has been extended from 3-dimensional to n-dimensional relationships. The usefulness of this new method for monitoring the impacts of toxicants to aquatic communities is demonstrated through examination of phytoplankton data for two oligotrophic lakes, one impacted by acid mine drainage and one pristine.

Abstract 35

BIOLOGICAL EFFECTS OF CONTAMINANTS IN HALIFAX HARBOUR SEDIMENT

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Halifax Harbour sediment is contaminated by heavy metals and organic contaminants. To assess the biological effects of the contaminants in this sediment, seven studied sites spanning a range of known sediment types, water depths, metal and organic contaminant levels were chosen for a series of bioassay tests. Sediment toxicity was measured by the reduction of microbial luminescence in three Microtox tests (pore water, solvent extract and solid phases), by the percent survival in two amphipod tests (*Rhepoxynius abronius* and *Corophium volutator*) and by the survival and change in biomass of juvenile polychaete (*Neanthes sp.*). Uptake of contaminants from sediment was assessed by the bioaccumulation test using a bivalve (*Macoma balthica*). The chronic effects of the sediment on vertebrates were assessed by histopathological studies in winter flounder (*Pseudopleuronectes americanus*) collected from the harbour.

The results of this study were used to evaluate the extent of contamination in Halifax Harbour sediment and to assess the potential value of a battery of laboratory sediment toxicity tests for marine sediment environmental assessment in the ocean disposal program administered by Environment Canada.

Abstract 59

TOXICOPATHIC HEPATIC LESIONS IN JUVENILES OF THREE SPECIES OF FLATFISH FROM PUGET SOUND, WA: RELATIONSHIP TO OTHER INDICATORS OF CONTAMINANT EXPOSURE.

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Liver neoplasms are rare in young wild fish; thus, one must consider other liver lesions as bioindicators of contaminant exposure effects in monitoring studies where target fish specimens are not adult. In addition, it has been argued that effects of contaminant exposure are more reliably assessed in juvenile fish which have not yet migrated extensively. We have addressed these issues by histologically examining juveniles of English and rock sole and starry flounder captured from nine sites in Puget Sound, and measuring fluorescent aromatic compounds (FACs) in bile and other biochemical indices of contaminant exposure. Although neoplastic and preneoplastic lesions were detected at low prevalences, much higher prevalences of several types of nonspecific and unique degenerative lesions were detected in all three species. These lesions have been experimentally induced in fish by exposure to various toxicants, and/or have been associated with contaminant exposure and the process of liver neoplasia in adult fish. Prevalences of these earlier bioindicators were significantly higher at the more contaminated sites compared to the less contaminated sites. Moreover, prevalences of these lesions in all three species were significantly correlated with mean bile FACs levels at the sites, in agreement with the results of previous studies utilizing adults. These findings further support the utility of certain liver lesions other than neoplasms as early indicators of biological damage in wild fish exposed to xenobiotics.

Abstract 63

THE RELATIONSHIP OF SEDIMENT CONTAMINATION TO THE OCCURRENCE OF PRENEOPLASTIC AND NEOPLASTIC LIVER LESIONS IN ENGLISH SOLE (*PAROPHRYS VETULUS*) FROM VANCOUVER HARBOUR.

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Three years of study have indicated the presence of preneoplastic and neoplastic liver lesions in English sole (*Parophrys vetulus*) from Vancouver Harbour. Compared to the absence of such lesions in English sole in Loughborough Inlet, a relatively undeveloped fjord along the B.C. coast, up to 75% of the fish in Vancouver Harbour were affected by these liver lesions. Highest prevalences were observed in Port Moody Arm, an area of the harbour which has the least tidal exchange and a history of oil refinery among other industrial and urban discharges. Concentrations of selected sediment contaminants, measured in the vicinity of the fish capture sites, were compared to the frequency of liver lesions in the sole. These included polycyclic aromatic hydrocarbons (PAH) which have been linked to fish liver lesions. Since commercial and sport fishing take place in Vancouver Harbour, the prevalence of preneoplasms and neoplasms in the fish raises questions concerning the environmental impacts and potential human health risks.

Abstract 67

ENHANCEMENTS TO NOAA'S STATUS AND TRENDS PROGRAM (NSTP).

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The NSTP has recently been expanded to allow better determination of the relationships between toxic chemicals and associated biological effects in coastal ecosystems. One area of enhancement has been the incorporation of more sensitive biological effects, such as xenobiotic-inducible enzyme activity in benthic fish species. The results from two years of measurement of isoenzymes of cytochrome P-450 (P450) or P450-dependent mixed function oxidase (MFO) activities show that this enzyme system is generally responsive to contaminant exposure in several species from both the East and West Coasts. However, additional studies (e.g. dose-response) are planned in order to better interpret the results. Another area of enhancement to the NSTP is the recent evaluation of reproductive dysfunction as a possible serious biological effect associated with exposure of benthic fish to toxic contaminants. In a series of field studies, we have shown that impaired ovarian maturation and failure to spawn can result from contaminant exposure, and we have found that levels of plasma estradiol and hepatic MFO activities are useful predictors of these effects.

Abstract 86

MONITORING ORGANIC CONTAMINANTS IN CANADIAN SEABIRDS, 1968-1990.

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Since 1968, eggs of selected seabird species have been collected at four-year intervals from colonies in eastern Canada and analyzed for organochlorines. Periodic surveys of organochlorines in seabird eggs have also been conducted at colonies on the Pacific coast and in the arctic. This monitoring program was established to provide data on contamination of the marine environment and possible implications for seabird health. DDE and PCBs declined significantly in all species from both the Bay of Fundy and the Strait of Georgia by the early 1980s, but levels now appear to have stabilized. generally DDE has declined more than PCBs. Dieldrin, oxychlordane, HCH and mirex levels have declined at some locations but were stable at others. HCB and heptachlor epoxide levels remained steady or increased depending on the species and location. All measured organochlorines increased or remained steady between the mid-1970s and mid-1980s in a resident arctic seabird, the ivory gull (*Pagophila eburnea*). GC/MS analysis of Leach's storm petrel eggs from Newfoundland in 1964 showed that toxaphene levels were greater than PCB levels, and had increased about two-fold over levels in eggs collected in 1968 and archived in a specimen bank.

Abstract 87

LINKING ENVIRONMENTAL INFORMATION TO DECISION-MAKING: THE ROLE OF MONITORING AND INDICATORS.

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To manage ecosystems and their resources on a sustainable basis, sound information must be available on which decisions can be based and evaluated. Environmental reporting - the timely delivery of understandable environmental information - is an important tool which can help communicate this information to decision-makers and the public. In recognition of this, Environment Canada is leading a federal effort to develop a national profile of environmental indicators.

This paper examines how environmental indicators can be used to focus environmental information and facilitate its use for decision-making and communication purposes. The indicator concept is introduced, and the basic building blocks required for indicator development - such as environmental monitoring and the use of reference thresholds - are discussed. Examples of indicators which have been used successfully, both economic and environmental, are presented. The ability of current data sets to allow for the development of marine environment indicators is reviewed and found wanting. Suggestions are made from an environmental indicator viewpoint regarding the development of a national marine monitoring program. The need for data which permit temporal and spatial analysis at national and regional scales is emphasized.

Abstract 88

MONITORING THE COASTAL AND ESTUARINE HEALTH OF THE UNITED STATES USING BIVALVE MOLLUSKS AS SENTINEL ORGANISMS.

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The U. S. National Oceanic and Atmospheric Administration uses bivalve mollusks to monitor national estuarine and coastal health. The project was initiated in 1986 and has collected mollusks and associated sediments annually since that time. Three mollusk species (*Mytilus edulis*, *Mytilus californianus*, and *Crassostrea virginica*) are used to quantify more than 70 contaminants along the conterminous United States and Alaska coasts while the use of an additional species (*Ostrea sandvicensis*) is necessary for the Hawaii islands.

With data available from 1986-1989 it can be concluded that concentrations of heavy metals and organic contaminants are highest in urban areas. Also, for certain estuaries both heavy metals and organic contaminant concentrations and associations are specific to the given estuary. Analysis of the data for possible temporal trends indicate that at least some contaminants may be decreasing at certain sites while increasing at others.

Abstract 89

MONITORING ALGAL BLOOMS CAUSING SHELLFISH POISONING AND FISH KILLS.

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Blooms of toxic species of algae can cause the death of marine fauna or harm humans when contaminated shellfish or fish are consumed. Several groups of algae and a variety of toxins, both water- and fat soluble, are involved and it is unlikely that any coastal area throughout the world is free from them. Aquaculture activities have contributed to an increase in the incidence of reported outbreaks, chiefly through the increased amount of product exposed to such blooms and widespread marketing, but there are concerns that the farming activities themselves could lead to the increase in blooms through eutrophication.

In British Columbia the chief problems are paralytic shellfish poisoning in both wild and cultivated bivalves, produced by several species of dinoflagellates of the genus *Protogonyaulax* (*-/Alexandrium*), kills of farmed salmonids by chloromonad flagellates (*Heterosigma* plus a new, undescribed form) and diatoms with barbed setae. Blooms of such organisms are natural events and cannot be prevented although their impact can be reduced through monitoring of the plankton and other key environmental parameters. The paper outlines some basic features of such programs, with particular reference to British Columbia.

Abstract 90

RIVERINE INPUTS IN MARINE ENVIRONMENTAL QUALITY MONITORING.

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It is important to understand the relative importance of riverine inputs when monitoring the environmental quality of coastal areas. Large data sets of riverine trace element concentrations exist; however, many of the data available are not suitable for determining either baseline conditions or water quality trends. Dissolved trace element data can be affected by the chemistry of the river under study, obscuring possible anthropogenic influences. Because many trace elements, particularly metals, are particle reactive, suspended particulate matter may provide a better basis for monitoring. Episodic, storm-related events are extremely important in the flux of materials through rivers. Such temporal variability in riverine trace element concentrations must be considered to understand riverine inputs. Lastly, in any monitoring program, the comparability of data from different laboratories must be assessed.

Abstract 91

STATISTICAL CONSTRAINTS OF STATUS AND TRENDS ANALYSIS OF ENVIRONMENTAL QUALITY DATA: TRENDS IN CONTAMINANT LEVELS IN FISH.

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Modern statistical methods of analysis offer assistance in the determination of contaminant trends in fish tissues. There are particularly useful in those instances where the rates of change are small. Two approaches will be described.

Analysis of covariance can be improved by the use of newly developed weighting methods for handling problems of inequalities of annual slopes of contaminant level and unequal variances.

Multivariate analysis of (co)variance offers a superior approach to determining trends when dependent variables are generally mutually correlated. A method of using principal components of correlated biological (independent) covariable in trend analysis will be outlined as applied to Canadian cod data.

Abstract 92

MARINE MAMMALS AS INDICATORS OF ENVIRONMENTAL CONTAMINATION BY PCBS AND DIOXINS/FURANS.

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Tissues of seals and whales (mainly blubber) have been used since the late 1960's to monitor PCBs, organochlorine (OC) pesticides and, recently, chlorinated dioxins/furans (PCDDs/PCDFs) in the marine environment. Until the late 1980's knowledge of these contaminants in Canadian marine mammals was largely confined to levels of PCBs (Aroclor) and DDT in seals from the East coast and the Arctic. We have recently completed surveys of a wider array of OC pesticides (toxaphene, chlordane, HCH isomers, PCBs and PCDDs/PCDFs in Arctic ringed seals (*Phoca hispida*), St. Lawrence and Arctic belugas (*Delphinapterus leucas*), killer whales (*Orcinus orca*) and porpoises from the B.C. coast. OC pesticides, especially toxaphene, and PCBs continue to be present at μg/g levels in Canadian marine mammal blubber especially in animals living near urban areas. St. Lawrence beluga (oil) had high levels of PCBs (85.5±63.3 μg/g in males) and mirex relative to those from the Arctic, but 2,3,7,8-TCDD was undetectable (<2 ng/kg) were present. The major PCDD congener in cetaceans from the B.C. coast was 1,2,3,6,7,8-HxCDD (<4 - 128 ng/kg). The presence of this congener suggests that use of chlorophenols or contaminated wood chips may be the source of PCDD contamination.

Abstract 93

FISH HEPATIC MONO-OXYGENASES IN MARINE ENVIRONMENTAL QUALITY ASSESSMENT.

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Fish hepatic mono-oxygenases (mixed function oxidases: MFO) are induced by a range of environmental contaminants including some polynuclear aromatic hydrocarbons (PAH), polychlorinated biphenyls (PCB) and chlorinated dibenzodioxins and dibenzofurans. Measurement of MFO activity therefore indicates the presence and effects of such compounds. In this paper, the use of MFO measurements to assess the scale and duration of pollution events, usually involving PAH, is reviewed. The success of MFO measurements in international exercises sponsored by IOC and ICES to evaluate methods of "biological effects monitoring" and the application of MFOs in future large scale marine environmental quality monitoring programmes will be discussed.

Abstract 94

OIL POLLUTION MONITORING - VALUE OF CHEMOMETRICS TO DATA QUALITY AND INTERPRETATION.

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With modern instrumental systems, particularly the "hyphenated" ones such as gas chromatography-mass spectrometry, it is relatively easy to generate a huge data set from a single determination. When tens or even hundreds of samples are analyzed, the researcher is faced with an apparent overload of information. Over the last couple of decades, however, mathematical and statistical techniques have been developed to extract the essential information from such chemical data sets. Indeed, this information can generally be presented in 2D or 3D plots that are easy to comprehend. Another virtue of the methods is that they are very sensitive to bad data points. Reinspection of data prompted by this sensitivity reveals misplaced decimal points, inverted numbers, peak misassignments and the like, thereby greatly enhancing data quality. An ongoing study of hydrocarbons and other compounds in the MacKenzie River and Estuary has successfully used chemometric tools to enhance data quality and understanding. This study will provide a sound basis for measuring the impact of future oil production in the southern Beaufort Sea.

Abstract 107

EVALUATION OF THE APPARENT EFFECTS THRESHOLD (AET) AS A BASIS FOR SETTING MARINE SEDIMENT QUALITY CRITERIA.

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The AET approach utilizes measured concentrations of contaminants in marine sediments and parallel sediment toxicity data to predict adverse biological effects. There is a strong impetus in the U.S. and in Canada to use the AET values as a basis for setting marine sediment quality criteria.

A recently completed research program for the American Petroleum Institute was designed to evaluate several aspects of the AET approach. Bioavailability and toxicity for a range of pollutants were evaluated in laboratory amphipod bioassays with a range of doses of contaminated sediments from Puget Sound. Despite the exceedance of the AETs for several metals and the observed bioaccumulation of certain PAHs, we observed no significant amphipod toxicity with the test sediments. These data suggest that sediment toxicity cannot by readily attributed to individual contaminants in complex mixtures.

Abstract 108

A MULTIVARIATE APPROACH FOR DEFINING SPATIAL IMPACTS USING THE SEDIMENT QUALITY TRAID.

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A concurrent assessment of benthic infaunal community structure, sediment chemistry, and sediment toxicity (known as the Sediment Quality Triad), was employed to develop an objective, multivariate approach to evaluate the spatial extent of benthic impact in an aquatic environment. Data acquired from 20 stations in Burrard Inlet, B.C. included 221 invertebrate taxa abundance estimates, 30 sediment chemical measures, and 7 toxicological responses. A Principal Components Analysis was performed on the two response attributes (community structure and sediment toxicity) to reduce their multidimensionality, then relate the observed spatial response patterns to the measured sediment levels for each chemical. Significant correlations $p<0.001$) for cadmium, copper, hydrocarbons, and sediment volatile residue were documented for spacial changes in benthic community structure. Few positive toxic responses were observed. Using simple linear regressions, the predicted benthic response for selected sediment quality criteria for cadmium and copper, were obtained. Predicted spatial impacts (i.e. criteria exceeded) were portrayed graphically. The feasibility of this approach for use in data management and for regulatory purposes was discussed.

Abstract 109

LONG-TERM EXPOSURE OF *NEANTHES* TO TOXIC MARINE SEDIMENT

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The effect of long-term exposure of juvenile *Neanthes arenaceodentata* to toxic marine sediments was examined to determine the relationship between changes in juvenile biomass and endpoints associated with reproductive success. At the end of a 25 day exposure period, the highest average individual biomass reported was for the West Beach (WB) control and Carr Inlet (CI) reference sediments, with the lowest biomass reported for Elliot Bay (EB). As the amount of contamination (determined by the EB fraction in the test sediment) increased, biomass decreased to almost zero growth in the most contaminated sediment (100 percent). Delay in reaching sexual maturity increased as the contamination increased. Worms exposed to WB and CI sediments matured between days 20 and 25, whereas worms exposed to the EB or an equal volume mixture of CI and EB sediments (CI/EB) matured between days 25 and 32. EB male and female worms exhibited significant mortality during the adult stage. During the 63 day period over which the experiment was conducted, one WB pair and 50 percent of the CI pairs produced egg cases. None of the CI/EB or EB pairs reproduced. The results of this study indicate that the level of contamination affecting juvenile growth in *Neanthes* is similar to the level that affects reproductive success.

Abstract 116

SUSTAINABLE DEVELOPMENT, ENVIRONMENTAL CAPACITY AND WATER QUALITY STANDARDS: INTRODUCTORY PERSPECTIVES.

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The concept of sustainable development has achieved a high public and political profile in recent years. It provides an over-arching goal and frame of reference that links environmental and economic imperatives and interests. Much work is ongoing to elaborate the policies, principles, and procedures for achieving sustainability. This research is being applied to various resource contexts, including aquatic and fisheries management.

In this paper, we explore the policy linkages between sustainable development, environmental capacity and water quality standards. The purpose is to provide a background/introductory perspective on contemporary issues of aquatic toxicity. We intend to argue these four points: 1)Environmental capacity is the enabling condition of sustainable development; 2) given present trends, the source and sink functions of most aquatic systems must be maintained at or near present levels; 3) conventional approaches to establishing water quality (or assimilative) standard and undertaking environmental assessment must be revamped; 4) a greater emphasis should be given to regional thresholds and the cumulative relationship between total stresses and eco-system integrity.

This brings into policy-focus some important scientific and technical issues regarding toxic discharges and their cumulative impact on aquatic and human health.

Abstract 117

A TOXICOLOGICAL PERSPECTIVE ON ECOSYSTEM CHARACTERISTICS TO TRACK SUSTAINABLE DEVELOPMENT.

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"Ecosystem Health", an emerging science paralleling human and veterinary medicine, has goals of systematic diagnosis and treatment of stressed ecosystems. Ecosystems are stressed by physical factors like boat traffic, biological factors like the introduction of an exotic species, and chemical factors like pH change. Even if these classes of stressors affect the same trophic levels, the resulting ecosystem disease states have different etiologies because the stress is introduced and propagated by different mechanisms.

This paper presents a toxicological perspective of ecosystem sustainability. I discuss how classical toxicological concepts have to be modified when the experimental unit is an ecosystem. A probabilistic definition of "ecosystem risk" (R_E) is given. The "health" of an ecosystem is then $1-R_E$. When exposure is high, effects are acute and are often measurable (e.g. fish kill). However, when exposures are low and chronic, effects are often hard to separate from the background. With these types of exposures, evidence of high risk for lack of sustainable development is exceedence of ecosystem "threshold criteria". Examples which use a systematic diagnostic framework illustrate the concepts.

Abstract 118

SUSTAINABILITY OF FISH POPULATIONS: IMPLICATIONS FOR TOXICOLOGY.

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Sustainable development implies development of human societies in a way the maintains populations and ecosystems in desirable states. For fish populations, sustainability implies maintenance of physical habitat, water quality, community structure, and population structure. Management for sustainability requires that we (1) define the essential characteristics of the systems we wish to preserve, and (2) determine how these characteristics respond to multiple environmental stresses with complex distributions in time and space. Independent management of single stresses (fishing, point-source effluents, nonpoint-source inputs) must be replaced by integrated management of all stresses. The past role of toxicology in environmental management has been limited to determining lethal or sublethal exposure levels for single toxic chemicals or effluent mixtures. Management for sustainability of populations will require integration of toxicological methods with management tools derived from fisheries science and ecology. This paper reviews progress that has recently been made in this direction and discusses changes in testing strategies required to fully integrate ecotoxicology into sustainable management schemes.

Abstract 119

MONITORING FOR SUSTAINABLE DEVELOPMENT.

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One of the key concerns regarding sustainable development is developing and implementing monitoring strategies that measure the state of ecosystems, resources, and conditions pertaining to human health. This ecosystem approach involves establishing monitoring goals consistent with sustainable development, then developing a sampling strategy to meet the monitoring goals. The process of monitoring for sustainable development is more complex than present monitoring programs. System attributes must be measured rather than simple characteristics or concentrations. Data collected must be useful to assess the probabilities of shifts to alternate ecosystem states and of remaining in the present state. Consequences of ecosystem state changes are evaluated with respect to the probability of changing to that state, i.e. the risk status. Risk assessment is the joint evaluation of the probability and consequences of a state change. To be successful, monitoring for sustainable development must provide information that will allow the prediction of changes in ecosystem state.

Abstract 120

WATER QUALITY GUIDELINES FOR SUSTAINABLE DEVELOPMENT.

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To realize the goals of sustainable development and bring environmental factors into the mainstream of decision-making, we must develop the scientific, environmental framework necessary to move sustainable development from a vaguely defined political philosophy to a practical, operational concept. Water Quality guidelines and objectives define realistic, scientifically defensible measures and goals of environmental sustainability that can be incorporated into the decision-making process alongside conventional economic and social criteria. Water quality guidelines and objectives are now widely used across Canada. The "goals and yardstick" value has potentially broad application to the sustainable development of water resources; i.e. environmental impact assessment, site rehabilitation, state of the environment reporting and the development of standards for aquatic ecosystem protection. With reference to Canadian programs, we discuss current trends in the development and application of water quality objectives in sustaining aquatic ecosystems. Topics include the development of water quality objectives for transboundary water management, the development of ecosystem objectives for Lake Ontario and the importance of public participation, and the move from tradition "end-of-pipe" approaches to environmental protection and regulation to approaches incorporating water quality and ecosystem objectives for receiving waters.

Abstract 128

SUBCELLULAR INDICES AS POTENTIAL ENVIRONMENTAL MONITORS.

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The use of subcellular markers for the determination of environmental stress from both metals and organics is reviewed. Evaluation of mixed function oxidase and specific metal binding proteins for metal stress are discussed. Data obtained from trout and oysters from B.C. are examined and the potential practical usefulness of these techniques is examined. Examples of MFO as applied to ground water runoff and the induction of metal binding proteins in fish exposed to mine tailings are presented. In addition, the relationships between species and biological effect are investigated.

Abstract A

THE EQUILIBRIUM PARTITIONING APPROACH FOR ASSESSING SEDIMENT QUALITY.

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Equilibrium Partitioning (EqP) sediment quality criteria (SQC) are the EPA's best recommendation of the concentration of a substance in sediment that will not unacceptably affect benthic organisms or their uses. EqP is being developed for the purpose of deriving National SQC for assessment and remediation activities. Early efforts considered a variety of methods to assess sediment quality. Recent review by EPA's Science Advisory Board of a variety of approaches that could be used to assess sediment quality has been completed with particular emphasis on EqP and AET methods. Although EqP SQC are similar to existing water quality criteria the application sediment criteria may vary significantly from the way water quality criteria are applied. It is not anticipated that the primary role of sediment criteria to be used as mandatory cleanup levels. Sediment criteria can be used as a means for predicting or identifying the degree and spatial extent of contaminated areas such that more informed regulatory decisions can be made.

Abstract B

SPIKED SEDIMENT BIOASSAYS AS AN APPROACH TO THE DEVELOPMENT OF SEDIMENT QUALITY GUIDELINES.

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Toxicological tests of marine sediments contaminated by PAH's showed toxicity changes relative to sediment stability and/or disturbance. The amphipod *Rhepoxynius abronius* was exposed to sediment undergoing no stability, stability over 10 to 35 days prior to testing, and 35 day stability followed by disturbance and immediate testing. Test results showed acute toxicity decreased with longer periods of stability, and stability followed by disturbance produced an acute toxicity which was similar to the non-stabilized result. One possible reason for these responses is that contaminants may become less available over time due to their decreasing concentration in pore-water. Previous work demonstrated that radiolabelled DDT spiked into sediment showed that pore-water concentrations of DDT reduced and gradually stabilized over a period of 30 to 40 days, and that the available toxic portions were related directly to pore-water concentrations. These tests indicate that sediment stability/disturbance is related to apparent contaminant availability, and the results of acute toxicity tests involving *R. abronius* must take this into consideration when sediment quality guidelines are developed.

Abstract C

DEVELOPMENT AND APPLICATIONS OF SEDIMENT QUALITY CRITERIA USING THE APPARENT EFFECTS THRESHOLD APPROACH.

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Development of sediment quality criteria is important to provide regulatory agencies and resource managers the basis for remediating and managing existing pollution problems, as well as preventing future contamination. The U. S. Environmental Protection Agency, Region 10 (EPA) and the Washington Dept. of Ecology (Ecology) have used the apparent effects threshold (AET) approach to develop proposed sediment quality criteria for Puget Sound. Ecology has incorporated the proposed criteria in their draft sediment management standards. The AET approach bases sediment quality criteria upon observed relationships between levels of sediment contamination and associated biological effects. Typically, development of sediment quality criteria using this approach relies upon a substantial database; however, once the database is developed it can generate criteria values for all contaminants associated with adverse biological effects. As the database is expanded, numbers may become more broadly applicable. The approach has already proved useful for making management decisions in relation to dredging and dredged materials disposal in Puget Sound, classifying marine sediments as clean or contaminated, guiding Superfund clean-up decisions and targeting point and nonpoint source control efforts.

Abstract D

SEDIMENT QUALITY TRIAD APPROACH.

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The determination of pollution-induced contamination by chemical substances known to be toxic in some circumstances is reasonably straight-forward. However, estimating the potential consequences, i.e., whether the particular types and levels of that contamination are actually degrading the environment, is not often as clear. The Sediment Quality Triad approach consists of collecting synoptic measures of sediment contamination: a) the concentrations of selected chemical substances, b) sediment toxicity using bioassays, and c) a measure of possible *in situ* biological effects, most commonly benthic infaunal community structure. The major advantage of the Triad approach is that each component of the Triad complements the limitations of the other two to provide a means of determining areas where pollution-induced degradation has occurred. This approach has been applied to data from a number of locations to identify degraded areas.

Abstract E

Sediment Quality Guidelines for the National Status and Trends Program.

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NOAA annually quantifies contaminants in sediments from many sites around the nation. Contaminant concentrations can be compared among sites, but these comparisons provide no estimate of the potential adverse toxicological effects of the chemicals. Data from the equilibrium-partitioning approach, the spiked-sediment bioassay approach and from several approaches that rely upon matching chemistry and biology data collected in the field were assembled and evaluated. The field-collected data were assembled from studies performed in many different geographic areas. Overall, the data from 85 reports were used to develop the guidelines. The concentrations at which effects were initially observed and the concentrations above which effects were frequently observed in the different studies were identified. Also, the relative degrees of confidence in the guidelines for the analytes were identified. This approach had the disadvantage of being very labour-intensive, but had the advantage of providing a framework of data within which to evaluate new sediment chemistry measurements. The data from many different studies were very much in agreement for some chemicals, while for some other chemicals there was very poor agreement. The guidelines were used to estimate the potential for toxicological effects at the monitoring sites.

Abstract F

An Approach to the Development of Sediment Quality Objectives for Burrard Inlet.

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Sediment quality objectives have been proposed for Burrard Inlet by the B.C. Ministry of Environment. The general principle followed in developing these was, if possible, to set the objective below the lowest measured AET from Puget Sound, while using data for B.C. reference sites to reflect local conditions. Data for the Puget Sound reference site were used to determine the relevance of the AET to actual Puget Sound Conditions. The objectives for PAHs were usually 10% of the lowest AET, while those for metals were always less then the AET, except for chromium and nickel. For both these metals, data for the Puget Sound reference site were two to five times higher than the AET. The objectives were set at the mean concentrations for relatively uncontaminated Burrard Inlet sites, which were higher than the AET, but below the Puget Sound reference site levels.

Abstract U

DESIGN OF THE GULF OF MAINE MARINE ENVIRONMENTAL QUALITY MONITORING PROGRAM.

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Under the direction of the five jurisdiction Gulf of Maine Council on the Marine Environment, a monitoring program was designed to meet three goals: provide information on the status, trends and sources of human health risks; provide information on the status, trends and sources of risks to ecosystem integrity; and provide this information in an appropriate and timely manner to managers. The program is designed to integrate and enhance numerous existing municipal, state, provincial and federal programs, including the NOAA Status and Trends Program, the FDA National Shellfish Sanitation Program, and the CWS Seabirds Contaminants Program in addition to existing effluent permitting programs. Information will be integrated through use of a personal computer-based link designed to use the existing data base management system in those agencies. Information interpretation will be completed by the Council's Working Group. Given that one of the options is accepted by the Council, a pilot project will be implemented as early as 1991.

Abstract V

LESSONS FROM U.S. NATIONAL AND REGIONAL MONITORING PROGRAMS.

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Long-term inconsistency and poor availability of data and supporting information, are two major problems inhibiting successful assessment of long-term pollution trends in coastal waters of the U.S. Large- scale geographical and long-term trends of contaminant concentrations in fish, shellfish and sediments were developed from recent and historical data collected from a dozen national and several hundred regional and state monitoring programs conducted aperiodically over the past 25 to 30 years. National monitoring programs, involving up to 200 sites, were underway between 1965 and 1972, 1976-78 and 1985-present under various agency sponsorship. State and regional monitoring programs complimented federal programs in some areas (Maryland, Texas, California) but other areas were neglected by both national and state agencies. Reconstructions revealed 100-fold declines of DDT and other pesticides on a national basis during the past 20 years, while PCB concentrations declined only near well-known major point sources. Regional analyses revealed no substantial changes in concentrations of most metals in biota, despite marked declines in inputs and sediment concentrations supporting the idea that metals have not been important contaminants of the sea coast.

Once a monitoring program begins, all effort should be made to continue it and to resist change even in the face of changing technology. To date, there remains no national or regional commitment to establishing, on a continual basis, data management systems that accommodate historical as well as current data and supporting information. It is not necessary that all such data be digitized--availability alone is half the battle!

Abstract W

A STUDY OF BENTHIC CONTAMINANTS IN VANCOUVER HARBOUR, B.C. TO ASSESS THE ENVIRONMENTAL QUALITY.

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In May of 1985, Environment Canada initiated a study to assess the benthic environmental quality of Vancouver Harbour, B.C. The main study objectives included the distribution of selected contaminants of the sediments and bottom-dwelling biota in the harbour; the identification of potential sources of urban and industrial contaminants; and the need for remedial measures to improve the environmental quality. A total of 88 stations for sediment and 11 trawl stations for biota have been sampled in Vancouver Harbour (from Point Atkinson to Port Moody Arm) from May 1985 to October 1989. Arsenic, cadmium, copper, chromium, iron, mercury, nickel, lead, zinc, and polycyclic aromatic hydrocarbons were among the chemical parameters quantified in the sediments and biota. Additional chemical parameters measured in the sediment included hydrocarbons, chlorophenols, and polychlorinated biphenyls (PCBs). Further investigations looked at the prevalence of liver lesions in English sole in association with chemical exposure.

Abstract P-9

ASSESSMENT OF THE USE OF POLYCHAETE ANNELIDS AS MONITORS OF MARINE ENVIRONMENTAL QUALITY.

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Recent (post-1978) literature pertaining to the application of polychaete annelids as species useful as indicators of marine environmental quality in Canada and elsewhere was reviewed. The role of correctly-identified species in assessing environmental quality and the application of this information to environmental quality standards was found to be effective but currently under-utilized. It is recommended that analysis of polychaete species assemblages be included when setting regulations and guidelines towards effective control of land-based pollutants in industrial effluent monitoring programs as well as in national marine status and trends programs.

MUSSELS AS BIOINDICATORS: A CASE STUDY OF TRIBUYLTIN EFFECTS IN SAN DIEGO BAY

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ABSTRACT

As part of a Navy research program to evaluate the environmental effects of tributyltin (TBT) antifouling coatings and develop *in-situ* field bioindicators, native juvenile mussels were transplanted in San Diego Bay between 1987-1990. Serial seasonal transplants and intensive chemical sampling have documented temporal and spatial variability in TBT and the effects of TBT on growth, bioaccumulation and survival that have not been previously reported. Establishing this variability and identifying site-specific factors affecting juvenile mussel growth in San Diego Bay have facilitated environmentally relevant predictions of threshold levels for TBT effects on mussel growth. This represents the initial calibration of the mussel bioindicator for assessing TBT effects in San Diego Bay and establishes a significant refinement in the use of mussels as biological indicators. Based on the data presented here, questions have been raised regarding the environmental significance of the data and monitoring strategies that may be applicable to all bioindicator responses and monitoring programs.

INTRODUCTION

The first tributyltin (TBT)-based antifouling coatings were developed in 1961 but were not used in significant quantities for at least 10 years (Stebbing, 1985; Champ and Lowenstein, 1987) after demonstrating superior performance. TBT antifouling coatings are effective for up to 5 years compared to only 2 years for copper.based paints. By 1980 the U.S. Navy proposed painting its fleet with TBT antifouling coatings. An environmental assessment predicted an annual savings of $150,000,000 in fuel consumption and hull cleaning expenses and no significant adverse environmental impacts (U.S. Navy, 1985). Fleet implementation of TBT antifouling coatings was suspended because adverse effects on non-target organisms were observed with increased usage of TBT antifouling coatings in several European countries (U.S. Environmental Protection Agency, 1986). The use of biological indicators has played a significant role in decisions by state, federal and international regulatory agencies that have restricted the use of TBT antifouling coatings in the 1980's.

History of Regulations

In 1982 France became the first country to adopt restrictions on the use of TBT antifouling coatings based primarily on biological responses, i.e., abnormal shell thickening in the Pacific oyster *Crassostrea gigas* (Alzieu, 1986). In the U.S restrictions were not imposed until 1986 after observations on shell thickening in were supported by environmental measurements of TBT and associated with other adverse biological responses in laboratory studies (Abel et al., 1987). The U S. Environmental Protection Agency (EPA) began a special review of TBT antifouling coatings in 1986 (U.S. Environmental Protection Agency, 1986), established an advisory in 1987 (U.S. Environmental Protection Agency, 1987) and Congress passed regulations in 1988 (U.S. Public Law, 1988). The State of California imposed restrictions on the use of TBT antifouling coatings in January 1988 (State of California, 1988). The EPA advisory concentration for protecting marine life is 10 ng/ℓ and in the State of California 6 ng/ℓ. Regulations established by the State of California seem to have been unduly influenced by observations of shell thickening in *C. gigas* made by California Mussel Watch (Stephenson et al., 1986; Smith et al., 1987; Stephenson et al., 1988). The EPA special review also contains a requirement for an extensive monitoring and research program that includes bioindicators and chemical measurements to evaluate the effectiveness of the restrictions on the use of TBT. Most monitoring programs have been based on analytical measurements of TBT in water, sediment and tissues that do not directly measure biological effects. In addition, temporal and spatial variability generally have not been adequately measured for seawater TBT concentrations and the effects of TBT.

Biological Indicators

Although TBT antifouling coatings are designed to affect organisms that attach to ship hulls, the potential for environmental effects is significant among many non-target organisms. Molluscs seem particularly susceptible. Abnormal shell thickening in the cultured Pacific oyster Crassostrea gigas was first observed in the U.S. shortly after its introduction in 1976 and correlated with suspended sediment in preliminary studies (Key et al., 1976). Shortly thereafter the French discovered a correlation with shell thickening and proximity to marinas and the first association with TBT antifouling coatings (Alzieu, 1986). The regulation of TBT by the French preceded most other countries largely because of shell thickening on cultured oysters and before conclusive scientific evidence against TBT had been gathered (Salazar and Champ, 1988). This is an example of a biological indicator detecting potentially adverse environmental impacts before chemical detection. A biological indicator is the use of a biological response to quantify the presence of toxic chemicals in water and their effects on the biological environment.

Using biological indicators to assess and regulate environmental effects is becoming standard. Bioindicators have several advantages over water sampling and community studies for quantifying environmental levels of contamination and predicting environmental impact. Bioindicators provide integrated information about environmental conditions and bioavailability that cannot be defined with chemical sampling. In the last decade many regulatory requirements have shifted from chemical to biological measurements including survival, growth and bioaccumulation (Chapman, 1983; Chapman and Long, 1983; Phillips and Segar, 1986; Parrish et al., 1988). There has also been a shift toward site-specific bioassays and in situ field testing. Waldock et al. (1990) make an important distinction between the use of biological indicators as detectors of environmental contamination by monitoring tissue accumulation versus their use as predictors of environmental impact bye measuring other biological responses. Our field transplant studies combined both approaches for a more integrated assessment.

Regulatory agencies must deal with the high degree of uncertainty in evaluating environmental effects and recommending appropriate environmental management practices, particularly with TBT (Salazar and Champ, 1988). There has been a tendency to emphasize bioindicators that are reportedly specific to TBT like shell thickening in *C. gigas*. Most of the evidence considered by the regulatory process to be most significant comes from bivalve molluscs. This has occurred for a variety of reasons. First, it is generally believed that molluscs are more sensitive than other animal groups to TBT, the primary toxic component of organotin antifouling paints. Second, many bivalves have a cosmopolitan distribution and are commonly maintained in the laboratory. Third, filter-feeding bivalves may be more susceptible to TBT due to their feeding strategies. Fourth, many bivalves have an economic importance in the commercial shellfish industry. In Europe, critical evidence was associated with shell thickening in oysters *C. gigas* (Alzieu, 1986; Waldock, 1986). In the U.S., the critical evidence was associated with three laboratory studies that reportedly demonstrated unacceptable effects on growth and development in oysters (*C. gigas*, *Ostrea edulis*) and clams (*Mercenaria mercenaria*) (U.S. Environmental Protection Agency, 1987).

Indicators of Contamination

Natural mussel populations have been used extensively as indicators of TBT contamination by measuring tissue concentrations of TBT in mussel tissues (Wade et al., 1988; Short and Sharp, 1989; Valkirs et al., 1990). Mussel transplants have been used similarly (Zuolian and Jensen, 1989). Bioaccumulation of TBT in mussel tissues is a potentially good indicator of environmental contamination because a single tissue measurement integrates long-term exposure to environmental concentrations of bioavailable TBT. Tissue measurements are therefore more environmentally meaningful than instantaneous, site-specific water measurements. However, bioavailable dose and actual exposure must be accurately quantified to definitively correlate water concentrations with tissue concentrations. Even then, accumulating levels of particular contaminants above ambient concentrations does not a priori indicate environmental impact to the bioindicator or other species (Peddicord, 1984). Using tissue accumulation in field bioindicators to quantify environmental levels of contamination is a potentially powerful tool, but there are many pitfalls involving interpretation and environmental significance (Phillips, 1980). Initial reports on mussels as bioaccumulators emphasized utility and minimized potential interference from extraneous environmental factors (Goldberg et al., 1978; Farrington et al., 1983; Goldberg et al., 1983). More recent reports have outlined potential problem in using tissue concentrations of contaminants for environmental prediction (Phillips, 1980; White, 1984; Phillips and Segar, 1986).

Indicators of Effects

Since bioaccumulation cannot be used to measure environmental effects directly, other biological responses (eg. growth, oxygen consumption, filtration rate) are used in laboratory and field tests but there are similar difficulties in predicting environmental impact from these data (White and Champ, 1983; Cairns and Buikema, 1984; Malins et al., 1984; Moller, 1987; Cairns, 1988). There is a tremendous gap between correlations and causality when using biological indicators in the field, extrapolating from the laboratory to the field, or comparing laboratory data with field data. Specific problem with the interpretation and environmental significance of TBT data have been discussed previously

(Stebbing, 1985; Salazar, 1986; Salazar and Champ, 1988; Salazar, 1989) as have the difficulties in extrapolating effects from shell thickening. Mussel growth represents the integration of biological responses to the environmental milieu, and reduced growth could have significant ecological consequences.

Both natural and pollution-related stresses have been shown to reduce mussel growth rates (Bayne et al., 1985). Reduced mussel growth has been associated with TBT in laboratory and field studies (Thain and Waldock, 1985; Stephenson et al., 1986; Salazar and Salazar, 1987; Stromgren and Bongard, 1987; Valkirs et al., 1987; Salazar and Salazar, 1988). Juvenile mussel growth was the most sensitive indicator of TBT measured in San Diego Bay microcosm experiments (Salazar and Salazar, 1987; Salazar et al., 1987). Threshold effects have been predicted from laboratory studies (Thain and Waldock, 1985; Thain, 1986, Thain and Waldock, 1986; Valkirs et al., 1987; Stromgren and Bongard, 1987), a portable flow-through field system in San Diego Bay (Salazar et al., 1987) and from mussel transplants in San Diego Bay (Stephenson et al., 1986; Salazar and Salazar, 1987, Salazar and Salazar, 1988; Salazar and Salazar, 1990a, Salazar and Salazar, 1990b). Most of the early laboratory studies and the earliest field study were conducted at very high test concentrations > 100 ng/ℓ, but since mean concentrations at all test sites in San Diego Bay are now < 100 ng/ℓ these early predictions are environmentally irrelevant. Juvenile mussels have particular advantages over adults as bioindicators: growth is not affected by gametogenesis (Rodhouse et al., 1986) and juveniles may be more sensitive to TBT than adults (Hall and Bushong, 1990). In addition, bioaccumulation in short-term tests with fast-growing juveniles more accurately reflects recent environmental changes (Fischer, 1983; Fischer, 1988).

U.S. Navy Monitoring Strategies

The Navy has a need to predict the environmental impact of a proposed program to paint their ships with TBT before implementation, and monitor effects thereafter. The Navy selected an integrated mussel monitoring approach because mussels are commonly used throughout the world and have been widely used in the U.S. and California. The objectives are similar to those of the California Mussel Watch program (Martin, 1985), i.e., to document and assess long-term trends in selected indicators of water quality, and provide data that are compatible with other monitoring programs. One part of the Navy monitoring program emphasizes many stations with quarterly monitoring of TBT in seawater, natural adult mussel tissues and sediment (Seligman et al., 1986; Seligman et al., 1990). This approach was part of a congressionally-mandated monitoring program whose main function was to document the extent of contamination through chemical measurements over time.

The work reported for our field-transplant studies is part of a Navy research program to develop biological indicators. We emphasize far fewer stations with more intensive serial sampling of TBT in seawater and mussel tissues as well as multiple biological measurements of transplanted juvenile mussels. Tissue TBT concentrations indicate the extent of contamination and mussel growth indicates environmental effects. Previously we discussed the use of mussels as bioindicators to determine the effects of TBT on survival, bioaccumulation and growth and established general relationships under natural conditions (Salazar and Salazar, 1990a). A model was developed to show the interactions between predicted threshold responses and various environmental variables. We have also discussed site-specific differences in TBT contamination and its effects and established temporal and spatial variability (Salazar and Salazar, 1990b). Predicted threshold responses were lowered based on site-specific effects.

Objective

The overall objective of this work was to examine the effects of TBT on mussels in San Diego Bay. More specific objectives included the following: 1) Document temporal and spatial variability in TBT concentrations and mussel responses; 2) Determine threshold response levels for concentrations of TBT in seawater and mussel tissues; 3) Calibrate and evaluate the utility of the mussel bioindicator. The main purpose of this paper is to discuss the environmental significance and monitoring implications of this work.

METHODS AND MATERIALS

Nine field-transplant studies were conducted for 12-week periods with natural populations of juvenile mussels (*Mytilus sp.*) in San Diego Bay between 1987 and 1990. One objective of these studies was to distinguish both the extent of TBT contamination and its effects among different sites. Intensive sampling over short time periods was conducted to detect differences. Serial seasonal transplant studies helped identify long-term trends. Details of these methods have been presented elsewhere (Salazar and Salazar, 1990a; Salazar and Salazar, 1990b).

Test Animals

Juvenile mussels (10-12 mm in length) were used to avoid the effects of gametogenesis on growth and because the literature suggested that juveniles are more sensitive than adults. Test mussels for these transplant studies were collected at the Magnetic Silencing Facility (MSF) just inside the mouth of the bay (Figure 1) where mean TBT concentrations in seawater and mussel tissues are among the lowest in San Diego Bay. Eighteen mussels were transplanted to each site at the beginning of each test. Test 1 animals were 10-15 mm in length (x = 12.0 mm). All others were 10-12 mm in length (x = 11.0 mm). Animals were continuously submerged either 1 m below the surface or 1 m above the bottom. Deep sites were included for comparative purposes because previous measurements in San Diego Bay marinas showed that seawater TBT concentration is of den lower near the bottom (Seligman et al., 1986; Seligman et al., 1990). The advantage of using transplants was the experimental control. Mussels can be transplanted to locations that require monitoring where they might not settle naturally. Effective settlement does not necessarily indicate healthy growth conditions. Another advantage of caged mussels is multiple measurements made on individuals throughout the test period.

Measurements

Mussels were measured and water samples collected weekly during Tests 1-4 and on alternate weeks (biweekly) during Tests 5-9 after it was determined that weekly measurements reduced mussel growth rates (Salazar and Salazar, 1990a). Water samples collected at those frequencies were measured for chlorophyll-α and TBT concentrations. Current speeds were measured occasionally with an in-situ meter. Whole animal wet weights, lengths, shell weights, tissue wet weights, and tissue TBT concentrations were measured at the end of each study. Only weight growth rates will be discussed because there is a greater range of response than with length growth rates (25x compared to 10x), length measurements are less accurate, and plateaus are reached more quickly during each 12-week test. A 12-week test length was selected because 1) previous bioaccumulation rate measurements with adult mussels suggested equilibrium was reached in about 60 days, and 2) this was the approximate limit before our 10-12 mm juveniles matured and gametogenesis would affect growth rates.

Test Sites

San Diego Bay was selected for these field-transplant studies because of the high concentrations of TBT found in marinas there, the large size of the Navy fleet homeported there and proposed use of TBT by the U.S. Navy. Current EPA regulations restrict use of. TBT antifouling coatings to vessels greater than 25 meters in length. However, due to widespread use on non-Navy vessels prior to 1988 and to vessels that still have TBT hull coatings, TBT persists in the environment at concentrations predicted to cause adverse effects. Monitoring sites were selected primarily for the greatest range of seawater TBT concentrations and other environmental conditions although access and security were also important. The 18 transplant sites extended from the northern to southern regions of San Diego Bay (Figure 1) and included seven marina sites and 11 Navy sites. Marina sites within enclosed yacht basins were selected because previous studies have shown that the highest concentrations of seawater TBT are generally associated with high densities of TBT-coated vessels in basins with poor flushing (Seligman et al., 1986). The relatively narrow entrances to these basins and other restrictions to flushing are shown in Figure 1. Navy sites were monitored because TBT antifouling coatings are not generally used on Navy vessels and previous monitoring studies indicated sediment and seawater TBT concentrations were quite low in these areas (Seligman et al., 1990). It was important to collect baseline information and use those data to predict environmental effects before the Navy could use TBT antifouling coatings.

The general relationships between seawater TBT, tissue TBT and growth rates have been discussed previously (Salazar and Salazar, 1990a) and comparisons between marina and Navy sites have been made (Salazar and Salazar, 1990b). To gain a better understanding of the threshold effects of TBT and the environmental significance of mussels as bioindicators, specific comparisons are made in this paper between two representative marina sites (SI, SID) and two Navy sites (NAV, MSF). The SI surface site (1 meter below the surface) and SID deep site (1 meter off the bottom) are located in the back of the Shelter Island Yacht Basin; a small area with very restricted flushing (Figure 1) that accommodates at least 2, 200 vessels. Water depth is approximately 5 meters. Seawater concentrations of TBT in this basin are currently the highest in San Diego Bay and were previously among the highest ever reported (Valkirs et al., 1986). These high seawater TBT concentrations have been shown to vary by more than an order-of--magnitude with tidal cycle (Clavell et al., 1986; Stang et al., 1989). The Magnetic Silencing Facility (MSF) is near the mouth of San Diego Bay It is characterized by

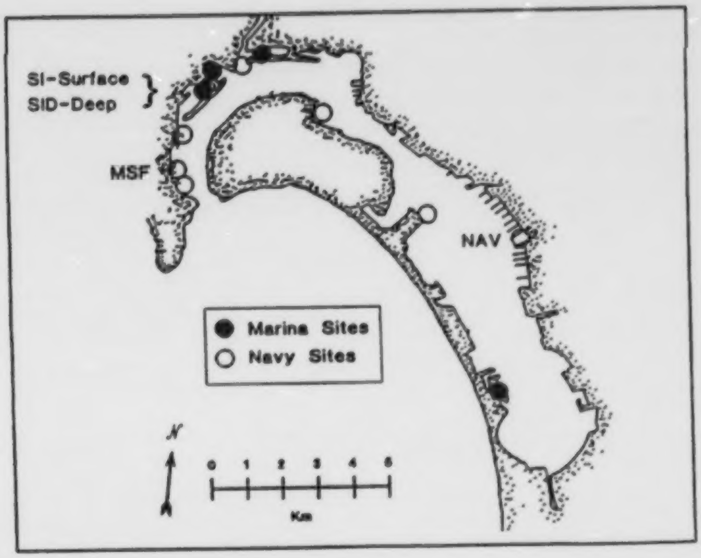

FIGURE 1. Juvenile mussel transplant sites in San Diego Bay. Marina (•) and Navy (0) sites are differentiated and four representative sites discussed in the test are called out. Two marina sites (SI, SID) are in the same location separated by 3 meters depth. The Navy sites are in the southern (NAV) and northern regions of the bay (MSF). MSF was also the mussel collection site.

good flushing, relatively uncontaminated water, coastal ocean temperatures and very few vessels. Water depth is approximately 5 meters and the site is influenced by both ocean and bay water. MSF is the site least influenced by activities in the bay and bay geography Naval Station San Diego (NAV), is in the southern end of San Diego Bay and is characterized by many large vessels adjacent to rows of long piers associated with high activity and many support operations. Flushing is suppressed, temperatures are elevated and chlorophyll-α concentrations are low. Water depth is approximately 12 meters. Although seawater TBT concentrations are low there, we have measured other contaminants at high concentrations. NAV is the most southern Navy site and exemplifies the extremes of bay conditions.

Data Analysis

Mean growth rates and seawater TBT concentrations for each site in each test were estimated from linear regressions on weekly or biweekly measurements. Each data point for mussel growth and tissue accumulation represents a 12--week mean for 18 animals. Graphical methods were used to display the general relationships among environmental levels of seawater TBT, tissue accumulation of TBT, and mussel growth. The statistical significance of each relationship was determined from linear regression analyses. Linear regressions were also used to determine the significance of long-term trends. In the time-series data, an Analysis of Variance (ANOVA) was used to determine differences between sites by test and across tests for seawater TBT concentrations, growth rate, temperature, and chlorophyll-α. Relationships were considered statistically significant at the 95% confidence level. Error bars represent ± two standard errors. Since tissue TBT concentrations are represented by single, end-of-test measurements, only differences across tests are compared statistically and no error term can be associated with individual test measurements. To demonstrate the extreme variability and differences in temperature, chlorophyll-α and seawater TBT concentrations at the two marina and two Navy sites, graphs have been produced from the individual data points rather than from mean values. Several comparisons were made between pooling all data, pooling data by region (marina vs Navy) and individual site data to assess the environmental significance and statistical validity of pooled data and ramifications for environmental monitoring programs.

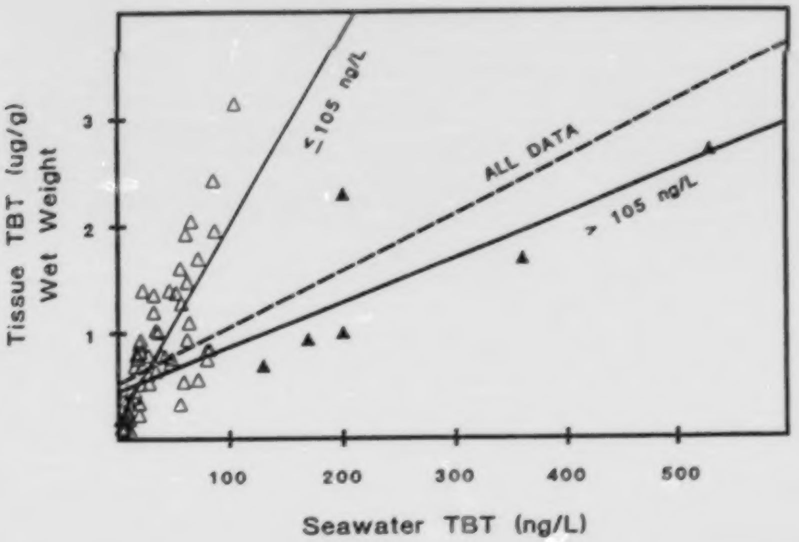

FIGURE 2. Positive linear relationship between TBT concentrations in seawater and mussel tissue. Seawater numbers are 12-week means and tissue numbers are end-of-test measurements. Regression lines show differences in tissue TBT accumulation between low (<105 ng/ℓ) and high (>105 ng/ℓ) seawater TBT concentrations and the regression for all data.

RESULTS

General Trends

Mean 12-week concentrations of seawater TBT ranged from 2 to 530 ng/ℓ and mean 12-week growth rates from 17 to 505 mg/wk (0.2 to 2.4 mm/wk). Transplanted juvenile mussels at every site accumulated measurable amounts of TBT during each test. Tissue concentrations of TBT ranged from 0.1 to 3.2 μg/g wet weight. The majority of mussels transplanted at marina sites accumulated TBT at concentrations >1 μg/g while mussels transplanted at Navy sites generally accumulated <0.5 μg TBT/g. These tissue TBT concentrations corresponded to seawater concentrations near 100 ng/ℓ and above in marinas and <10 ng/ℓ at Navy sites. Only at Navy sites adjacent to marinas were mussel tissue TBT concentrations >0.5 μg/g. The relationship between seawater TBT and growth is better than that for tissue TBT and growth. The lowest growth rates were associated with the highest concentrations of seawater TBT, temperatures near 14.5°C, and low chlorophyll-α concentrations. The highest growth rates were associated with low concentrations of TBT, high chlorophyll-α concentrations and low temperatures near 20°C. There are significant relationships between seawater TBT and tissue TBT. Growth rate is significantly related to both seawater TBT and tissue TBT. There is a statistically significant difference in seawater TBT, tissue TBT and mussed growth rate when marina sites are compared to Navy sites.

Seawater TBT & Tissue TBT

There is a significant linear relationship between seawater TBT and tissue TBT (Figure 2). However, this relationship is different for lower seawater TBT concentrations than higher ones. The slope of the regression for seawater TBT concentrations ≥105 ng/ℓ is almost 5 times higher than the slope at seawater TBT concentrations >105 ng/ℓ suggests a higher bioconcentration factor at lower seawater TBT concentrations. The highest seawater and tissue concentrations of TBT were found in marinas. The majority of seawater TBT concentrations were <100 ng/ℓ and tissue TBT concentrations <1 μg/g wet weight. Six of the seven highest seawater and tissue TBT concentrations were measured in the Shelter Island Yacht Basin (SI, SID). These data strongly influence the relationship between seawater TBT and tissue TBT shown in Figure 2.

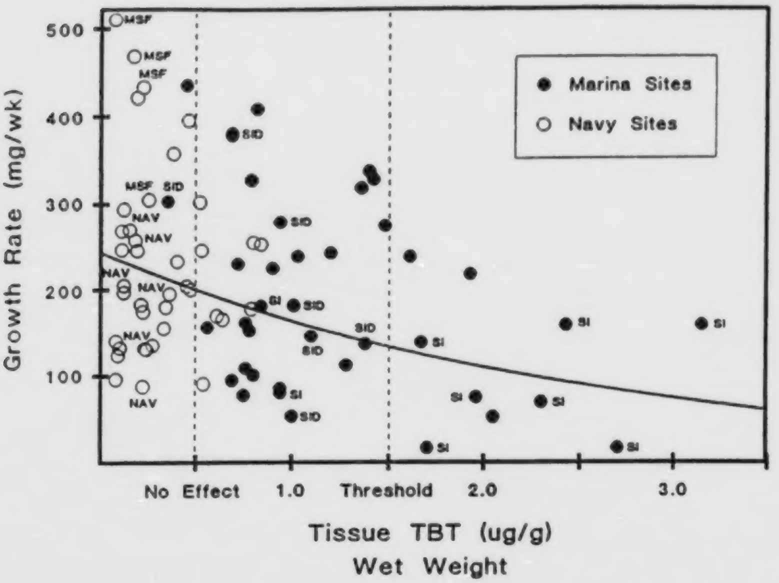

FIGURE 3. Negative exponential relationship between tissue TPT concentration and juvenile mussel growth rate. Marina (•) and Navy (○) sites are differentiated and the four representative sites discussed in the text are identified (SI, SID, MSF, NA). Predicted No Effect and Threshold Effect concentrations are shown by the dotted vertical lines.

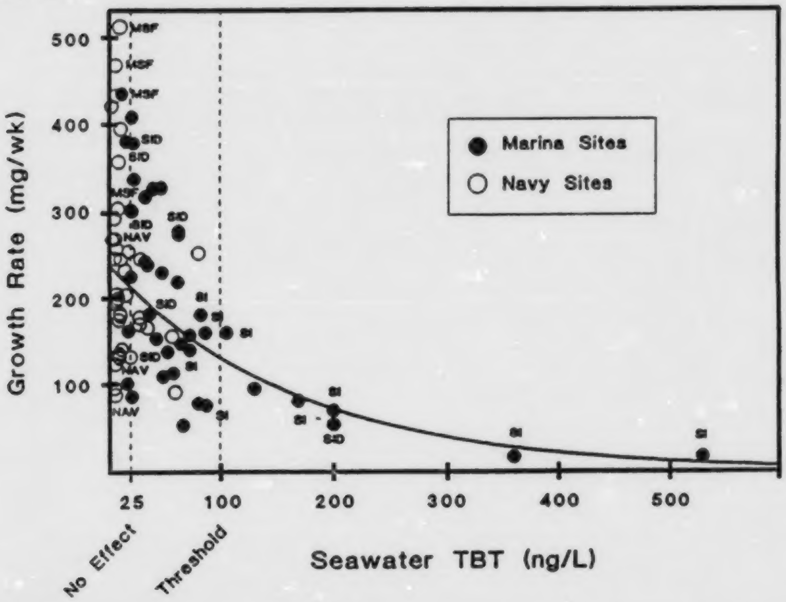

FIGURE 4. Negative exponential relationship between seawater TPT concentration and juvenile mussel growth rate. Marina (•) and Navy (○) sites are differentiated and the four representative sites discussed in the text are identified (SI, SID, MSF, NA). Predicted No Effect and Threshold Effect concentrations are shown by the dotted vertical lines.

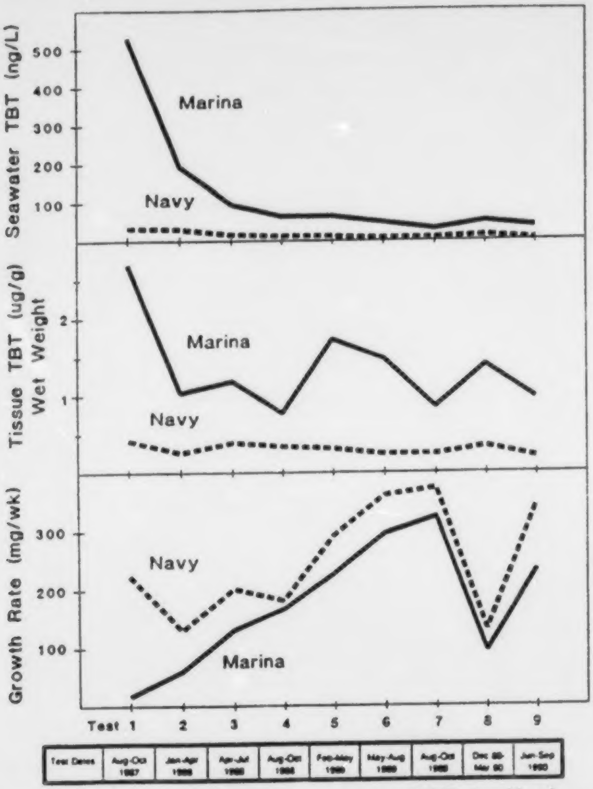

Figure 5. Temporal and spatial differences in mean seawater TBT (ng/ℓ), tissue TBT (μg/g) wet weight, and growth weight (mg/wk) for marina (—) and Navy (----) sites. Also given are test number and corresponding dates.

TBT & Mussel Growth

There is a significant negative exponential relationship between juvenile mussel growth and tissue TBT concentration. Figure 3 shows the high degree of variability in growth at all tissue TBT concentrations. Based on regression analyses, juvenile mussel growth rate is less dependent on tissue TBT concentration than seawater TBT concentration. Only 14% of the variance in growth can beexplained by tissue TBT concentration. This suggests that the toxicity of TBT in seawater has a more direct effect on mussel growth than TBT accumulated in mussel tissues. There are no significant regressions when the analyses are limited to data <1.5 μg/g. Based on analytical measurements, field observations and statistical analysis, the threshold level of tissue TBT concentration for probable effects on mussel growth was estimated at 1.5 μg/g and the no-effect level at 0.5 μg/g. There is a zone of uncertain effects between.

There is a significant negative exponential relationship between juvenile mussel growth and seawater TBT concentration (Figure 4). Based on regression analysis, 51% of the growth variance can be explained by seawater TBT concentration. Six of the seven highest seawater TBT measurements and five of the six lowest growth rates were measured in the Shelter Island Yacht Basin (SI, SID). As seawater TBT concentrations decreased, mussel growth rates for mussels transplanted at SID were among the highest measured at all sites. The seven data points for seawater TBT concentrations >100 ng/ℓ strongly influence the significance of the regression for all data. However, the equations describing the relationships for growth rate at high and low seawater concentrations of TBT are quite similar. There is also a significant exponential relationship when the seawater TBT data <100 ng/ℓ are analyzed separately, but in this case only 11% of the growth variance can be explained by seawater TBT concentration. The lowest seawater TBT concentration that resulted in a statistically significant relationship was near 70 ng/ℓ.

Environmental effects are uncertain, however, since only 7% of the growth variance can be explained by seawater TBT concentrations. Under the most adverse conditions, growth could be affected by lower concentrations. Based on analytical measurements, field observations and statistical analysis, the threshold level of seawater TBT concentration for probable effects on mussel growth was estimated at 100 ng/ℓ and the no-effect level at 25 ng/ℓ. There is a zone of uncertain effects between.

TBT & Effects: Marinas vs Navy

There is a statistically significant difference in seawater TBT, tissue TBT and mussel growth rates across tests when marina sites are compared to Navy sites. Figure 8 shows that the mean concentration of seawater TBT in marinas has declined rapidly since 1987 and by October, 1989, approached the mean for Navy sites. Seawater TBT and tissue TBT concentrations at marina and Navy sites were significantly different in every tests and growth rates were significantly different in most. Pooling sites suggests that seawater and tissue TBT concentrations decreased while mussel growth rates increased over time. However neither tissue TBT concentrations or growth rates consistently tracked seawater TBT concentrations. Tissue TBT concentrations were highly variable and growth rates increased steadily until Test 8 when a sharp decline was associated with winter seawater temperatures < 15° C.

TBT & Effects: Marina Sites

The lowest growth rates measured in San Diego Bay were 17 mg/wk at SI in both Tests 1 and 2. There is a statistically significant difference in seawater TBT concentration, tissue TBT concentration, and juvenile mussel growth rates when SI is compared to SID (Figure 6), These sites, located in the Shelter Island Yacht Basin, are separated by a vertical distance of only 3 meters. There was a dramatic decrease in seawater TBT at both sites from 1987 to 1990. Seawater TBT concentrations at the surface site (SI) were always higher than at the bottom site (SID). The seawater TBT concentrations at SI are approaching those at SID, but they are still significantly different. Tissue TBT concentrations did not consistently follow seawater TBT concentrations, particularly at SI where seawater TBT concentrations were the highest. Growth rates for mussels transplanted at SID were significantly higher than at SI. After Test 4 growth rates at SID were significantly higher than for mussels transplanted to most Navy sites, even though seawater TBT concentrations were significantly higher at SID. These individual site comparisons suggest that while seawater TBT concentrations decreased and growth rates increased, tissue TBT concentrations did not change significantly.

TBT & Effects: Navy Sites

NAV is the only Navy site that has been monitored in every test. At this site seawater TBT concentration has decreased, tissue TBT concentration has increased, and growth rate followed a seasonal cycle (Figure 7). The MSF site was monitored from Tests 5 through 9. During this time seawater TBT concentration decreased, tissue TBT concentration decreased and growth rates also followed a seasonal cycle. The highest juvenile mussel growth rate we ever measured in San Diego Bay, 505 mg/wk, was recorded at MSF in Test 9. In the tests where NAV and MSF could be compared, there were no statistically significant differences in seawater TBT concentrations or tissue TBT concentrations. There was a statistically significant difference in juvenile mussel growth rate. Seawater TBT concentrations and tissue TBT concentrations were much lower at the Navy sites than at the marina sites. Growth rates were also higher for mussels transplanted at the Navy sites, except during Tests 5-9 when growth rates for mussels transplanted at SID were higher than those at NAV.

Natural Variability in Seawater TBT

Some general trends in environmental concentrations of TBT, natural factors and their association with mussel growth can be more easily identified by pooling data for all stations by test, even though this procedure may be statistically inappropriate. The seasonal variability in temperature, chlorophyll-α and growth rate, and their relationship to seawater TBT concentrations and tissue TBT concentrations are shown in Figure 8. here was a rapid decline in seawater TBT concentration during the first three tests that was probably unrelated to seasonal factors. This was followed by a more gradual decrease in seawater TBT concentration with some intermittent increases that could be related to seasonal factors. Tissue TBT concentration seems to be associated with seasonal factors and mussel growth rate in addition to seawater TBT concentrations. There are seasonal cycles of chlorophyll-α, growth and temperature which covary by test and season. The lowest growth rates are associated with the lowest temperatures and the lowest concentrations of chlorophyll-α.

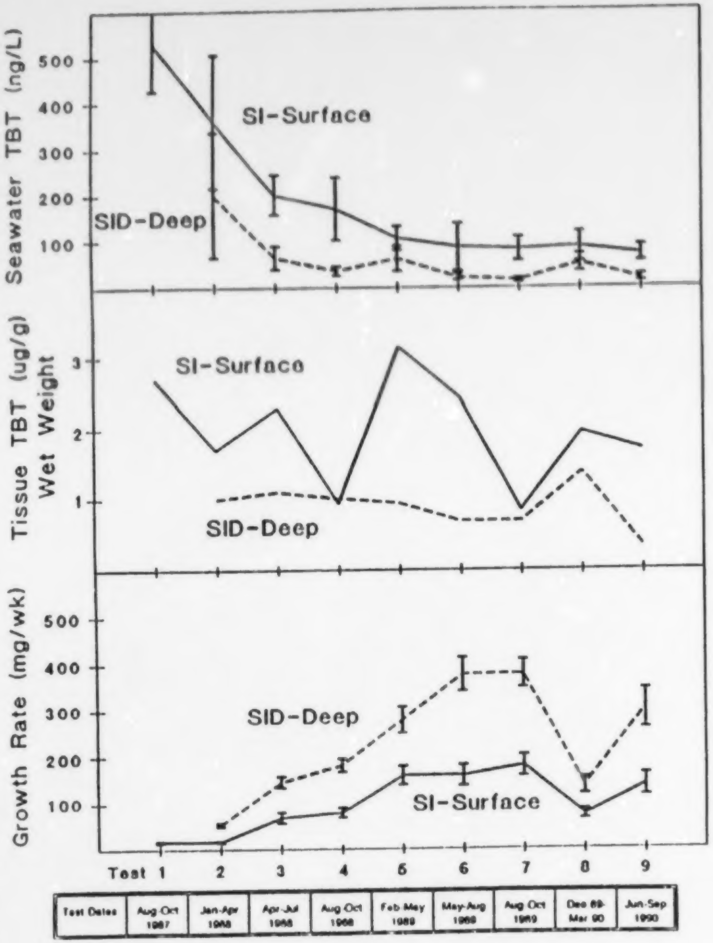

Figure 6. Temporal and spatial differences in mean seawater TBT (ng/ℓ), tissue TBT (µg/g) wet weight, and growth rate (mg/wk) for marina sites SI (——) and SID (- - -) in the Shelter Island Yacht Basin. These sites are separated by only 3 meters depth. Test numbers and corresponding dates are also given. Error bars represent ± 2 standard errors.

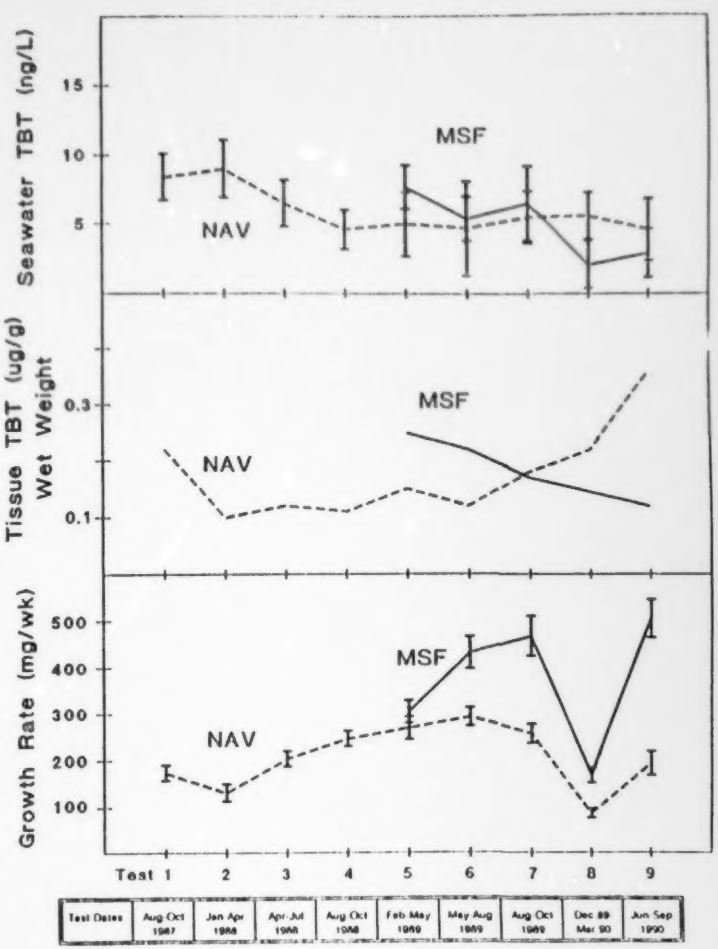

Figure 7. Temporal and spatial differences in mean seawater TBT (ng/∫), tissue TBT (µg/g) wet weight, and growth rate for Navy sites MSF (——) in the north bay and NAV (- - -)(Naval Station San Diego) in the south bay. Test numbers and corresponding dates are also given. Error bars represent ± 2 standard errors.

The fine structure of this temporal and spatial variability is shown in Figure 9 where the two marina sites (SI, SID) and two Navy sites (MSF, NAV, are compared with respect to seawater TBT concentrations, chlorophyll-α and temperature. The main purpose of this figure is to demonstrate the tremendous variability in these three factors over time but overall differences are also apparent. Although we found a statistically significant difference across tests between SI and SID temperatures, they are very close and the large number of data points largely obscure the subtle differences we found. It may be inappropriate to use mean values for these factors because it is not clear whether mussels are responding to means or extremes. At high seawater TBT concentrations we measured variability approaching an order-of-magnitude during the 12-week exposure periods. Chlorophyll-α was generally much lower in the winter than in the spring and summer. Chlorophyll-α concentrations were significantly higher at SID than SI, and significantly higher at MSF than NAV. Temperatures at NAV were generally much higher than MSF, but they were similar during the winter. Daily temperature variation was much greater at MSF than NAV. Similar differences in temperature variability were found between SI and SID, albeit not as dramatic.

There are three important differences in these parameters to be distinguished here that may have biological relevance. They are mean values, daily variability and seasonal changes. Mean values show statistically significant differences in seawater TBT concentrations between SI and SID but no differences between NAV and MSF. Mean values show statistically significant differences in both chlorophyll-α and temperature when comparing SI and SID as well as between NAV and MSF. Although seasonal variability is similar for each parameter and each site, the differences in daily or short-term variability are dramatic. The most dramatic differences were detected with *in-situ* temperature monitors that recorded temperature at half-hour intervals during the entire test period. Although meantemperatures are significantly higher at NAV, short-term variability at MSF was significantly higher.

DISCUSSION

We will discuss long-term trends in seawater TBT and tissue TBT and their effects on mussel growth, threshold responses predicted from these data, the environmental significance of short-term variability in natural factors, and the monitoring implications of the work. Several comparisons will be made to the Navy statutory monitoring program (Seligman et al., 1990, Valkirs et al., 1990). It should be noted that our bioindicator program and the monitoring program were both sponsored by the Navy and conducted in the same laboratory at the Naval Ocean Systems Centre, therefore all the measurement techniques are comparable. The purpose of our program was to develop an *in-situ* field bioindicator system to assess the environmental impact of Navy contaminants. We measured survival, growth and bioaccumulation of TBT in transplanted juvenile mussels to establish the extent of contamination and resulting effects. The monitoring program measured TBT in water, sediment and natural adult mussel tissues to establish an environmental baseline for TBT and document long-term trends.

Marina vs Navy Sites

A statistically significant difference in seawater TBT concentration, tissue TBT concentration and mussel growth rates was found for the test period 1987-1990 when Navy sites were compared with marina sites. The decrease in seawater and tissue TBT concentrations for both marina and Navy sites was similar to that reported by Valkirs et al. (Valkirs et al., 1990) for Navy statutory monitoring during the same time period. It is instructive to emphasize the similarities and differences since the monitoring program sampled ≈ 25 sites on a quarterly basis, whereas in our bioindicator approach we sampled ≈ 10 sites per test either weekly or biweekly for periods of 12 consecutive weeks over the 3-year test period. Both approaches produced similar results for pooled regions, but our intensive short-term sampling and use of transplanted mussels permitted site-specific distinctions to be identified that are not possible with long-term monitoring of natural populations. The precipitous decline in seawater TBT concentrations at marinas coincides with restrictions on the use of TBT antifouling coatings by the State of California in January, 1988 (State of California, 1988) and demonstrates the effectiveness of those restrictions. A similar but less dramatic decrease in seawater TBT concentration for Navy sites suggests that marinas are the primary source of TBT at all Navy sites and for the entire bay as previously reported by Seligman et al. (1986, 1990).

General trends can be misleading however, and pooling data from different monitoring stations can mask important differences and bias the mean values. Although tissue TBT concentrations in our study decreased at marina and Navy sites when pooled, some sites did not change significantly while others either increased or decreased in tissue TBT concentrations. These apparent anomalies suggest that factors other than seawater TBT concentration may be affecting TBT uptake in mussels. Increases in juvenile mussel growth rates were considered significant and related

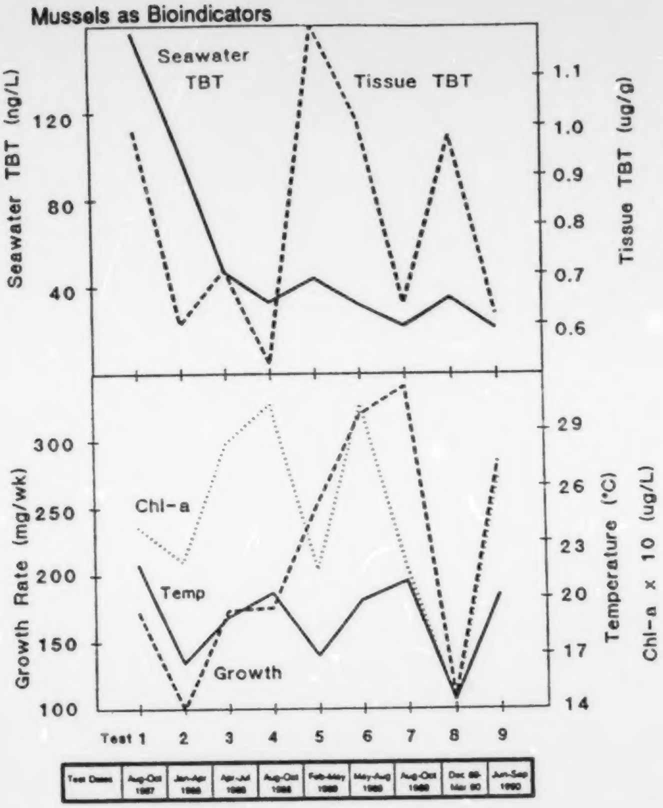

FIGURE 8. Mean seasonal variability in seawater temperature (°C), chlorophyll-a (µg/g) and juvenile mussel growth in San Diego Bay between 1987 - 1990. Mean seawater TBT (ng/ℓ) and tissue TBT (µg/g) wet weight are also shown.

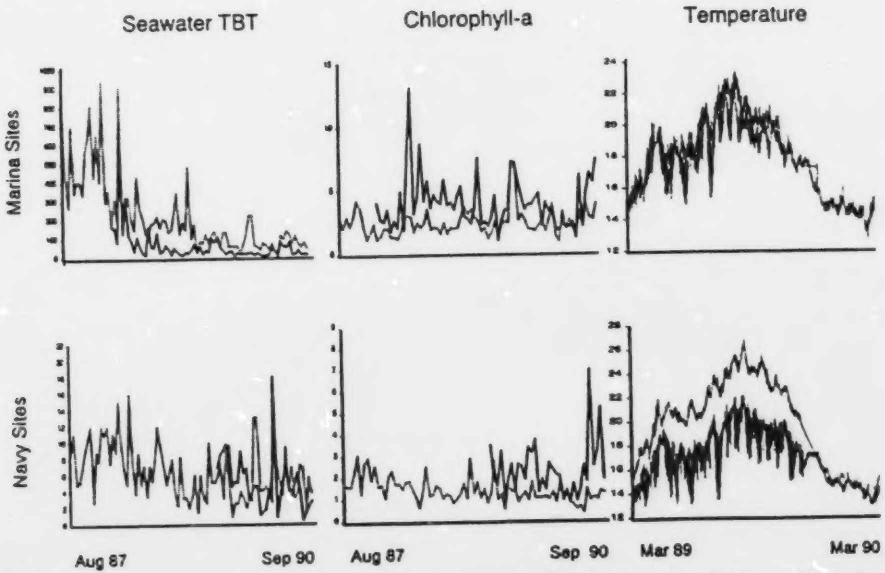

FIGURE 9. Comparison of temporal and spatial variability in seawater TBT (ng/ℓ), chlorophyll-a (µg/g), and temperature (°C) between two representative marina sites (SI - - -, SID —) and two representative Navy sites (NAV - - -, MSF —) between August 1987 – September 1990 (March 1989 – March 1990 for temperature).

to lower seawater TBT concentrations and reduced handling as previously reported (Salazar and Salazar, 1990a). We have also suggested how growth rates can affect bioaccumulation and therefore natural factors that affect growth rates can indirectly affect the bioaccumulation process.

Examining these trends in seawater and tissue TBT concentrations and mussel growth rates on a site-by-site basis reveals the fine structure of variability and helps define the advantages and limitations of this approach for environmental prediction. Although the statutory monitoring program gives a better overall chemical description of San Diego Bay by monitoring 25 sites within four regions, a single quarterly sample per site has no associated error term and cannot define short-term variability at a given site. We are able to associate an error term estimating variability in both seawater TBT concentrations and mussel growth rate with respective 12-week means and establish confidence limits on that number. Additionally, only the growth data provide a direct measure of environmental effects.

Marina Sites: SI vs SID

There are statistically significant differences in seawater TBT concentrations, tissue TBT concentrations and mussel growth rates in every test where the Shelter Island Yacht Basin surface (SI) and deep (SID) sites are compared. This is surprising since these two sites are separated by a vertical distance of only 3 meters. The differences show how pooling data for these sites could be misleading. The higher concentrations of both seawater and tissue TBT at the SI site (1 meter below the surface) compared to the SID site (1 meter off the bottom) also imply that ship hulls are the primary source of seawater TBT rather than TBT desorbed from contaminated sediment. This is consistent with sediment desorption measurements in San Diego Bay (Kram et al., 1989) and bioassays performed on TBT contaminated sediment from Commercial Basin (Salazar and Salazar, 1985). This particular sediment was the most contaminated in San Diego Bay (Valkirs et al., 1990) and yet there was no apparent toxicity from the very high levels of TBT measured in the sediment.

Higher growth rates suggest that mussels transplanted to SID near the bottom remained healthier than those transplanted near the surface. In addition to seawater TBT concentrations being lower at the bottom, water temperatures were lower and chlorophyll-α concentrations were higher. The added effect of these environmental factors increases the difficulty in separating the beneficial effects of low summer temperature and better nutrition from the adverse effects of TBT. A seasonal thermocline has been identified in the Shelter Island Yacht Basin (Stang et al., 1989) resulting in two very different water masses that could account for the differences in seawater TBT concentrations measured here and reported by others (Seligman et al., 1986; Stang et al., 1989). Tidal influence in San Diego Bay is very strong and order-of-magnitude changes in seawater TBT concentrations occur with the tidal cycle in the vicinity of the Shelter Island Yacht Basin (Clavell et al., 1986; Stang et al., 1989). Similar variability with tidal cycle has been reported for other contaminants as well (Zirino et al., 1978). All of these findings identify the surface site SI as being environmentally different than the bottom site SID. Such physical, chemical and biological differences argue against pooling data. They also show that additional physical and chemical parameters must be measured in monitoring studies to quantify environmental factors that could account for observed differences in juvenile mussel growth and bioaccumulation.

Navy Sites: NAV vs MSF

There were no statistically significant differences in seawater TBT concentrations and tissue TBT concentrations when the Navy sites NAV and MSF were compared. However, juvenile mussel growth rates for NAV transplants are significantly lower than growth rates for MSF transplants. Seawater TBT concentration was not a significant factor in regulating mussel growth rates at Navy sites because seawater TBT concentrations at most Navy sites were < 10 ng/ℓ. The physical, chemical and biological conditions present at each of these sites were quite different and probably responsible for the observed differences in juvenile mussel growth. MSF is near the mouth of the bay where flushing is good and concentrations of most contaminants are low. NAV is located in the southern portion of San Diego Bay. It is characterized by high summer temperatures near 25°C and low chlorophyll-α concentrations near 1 μg/g. We have also measured high concentrations of heavy metals and petroleum hydrocarbons in both seawater and mussel tissues at NAV. This is not surprising since the Naval Station supports many large vessels. This also suggests why chemical monitoring of a single contaminant should not be used for environmental prediction.

Further, chemical monitoring of TBT in seawater, sediment, and mussel tissues does not identify natural environmental stress factors or stress from other contaminants. Biological effects can only be inferred by comparing environmental levels of TBT with levels found to cause effects in laboratory studies. Biological monitoring can indicate effects without identifying the source. This has significant ramifications for the use of monitoring programs to determine threshold responses of contaminants and predict environmental significance. To determine the actual effects of TBT, other factors must be measured and evaluated before using correlations as cause-effect relationships. NAV (Naval Station San Diego) is a particularly important site because a high density of large vessels coated with TBT would be located here if application was approved. Since the chemical sampling of the statutory monitoring program did not measure biological effects and since we only had one Naval Station site (NAV), neither monitoring program can accurately document the biological environment at Naval Station San Diego or predict the environmental impact of coating the Navy ships there with TBT. These two monitoring programs could be combined and expanded to monitor other factors that would facilitate more accurate environmental prediction.

Threshold Responses

Seawater TBT, tissue TBT and mussel growth rates have been used to distinguish site-specific differences in contamination and effects (Salazar and Salazar, 1990b). Here, we have examined these relationships in more detail. Based on mussel growth rates and other factors, the threshold level of tissue TBT concentration for probable adverse effects on mussel growth is estimated at 1.5 μg/g and the no effect level at 0.5 μg/g. Between is a zone of uncertain effects. This threshold tissue TBT level is similar to one predicted from scope-for-growth studies (Page and Widdows, 1990) and oyster growth studies (Waldock et al., 1990). The threshold response level for seawater TBT is predicted at 100 ng/ℓ and the no effect level at 25 ng/ℓ. Between is a zone of uncertain effects. The threshold level is similar to the threshold predicted for effects on oyster growth. The no effect level is similar to the original regulatory level established of 20 ng/ℓ in the U.K. (Abel et al., 1986) and the EPA advisory of 10 ng/ℓ in the U.S. (U.S. Environmental Protection Agency, 1986). Although our data can be used to support these regulatory levels, we still believe the regulatory process is flawed (Salazar and Champ, 1988), and regulators may have made the right decision for the wrong reasons by using laboratory toxicity tests alone to establish regulatory criteria (White and Champ, 1983). We have demonstrated that restrictions on the use of TBT antifouling coatings have improved mussel growth in San Diego Bay, particularly in marinas and areas adjacent to marinas. This suggests that the field bioindicator approach is valid and that the regulatory process should include a similar field validation approach.

Mussel growth can be an effective and dynamic indicator of environmental stress because it represents an integrated response of biological processes to the environment. The first reported effects of TBT on juvenile mussel growth at seawater TBT concentrations < 100 ng/ℓ were recorded in our flow-through microcosm in San Diego Bay (Salazar et al., 1987). Although we found statistically significant effects at seawater TBT concentrations near 70 ng/ℓ, it was suggested that threshold effects were overestimated based on temperature and nutritive stress induced by the test system. After the first three San Diego Bay juvenile mussel transplant studies, significant effects of TBT on mussel growth were reported at seawater concentrations > 200 ng/ℓ and it was suggested that natural factors modified TBT effects at concentrations < 100 ng/ℓ (Salazar and Salazar, 1988). The general relationships from nine transplant studies also estimate threshold effects at 100 ng/ℓ (Salazar and Salazar, 1990a). It was acknowledged that effects could occur at lower concentrations if the animals were under stress from other environmental factors. Statistically significant effects were found near 70 ng/ℓ in this study as well, but the prediction of these effects is highly uncertain when extrapolating to natural conditions. The site-specific differences in our most recent work allowed detection of no effect levels and threshold levels for effects of seawater TBT and tissue TBT (Salazar and Salazar, 1990b). We have shown how the general relationships previously established by statistically significant regressions can be misleading unless sufficient site-specific data are reported. The threshold values for effects of seawater TBT and tissue TBT concentrations on juvenile mussel growth rates were adjusted downward to reflect site-specific effects. In other words, we associated an error term with the predicted thresholds. Statistical significance often differs from environmental significance. It appears that growth is a response to site-specific effects that incorporate the effect of extreme conditions as well as long term means.

There is a significant positive linear relationship between seawater TBT and tissue TBT concentrations but the relationship changes with seawater TBT concentrations > 100 ng/ℓ. The slope of the linear regression is approximately 5 times greater when only seawater TBT concentrations < 100 ng/ℓ are used. We have previously suggested that seawater TBT concentrations could not be predicted from tissue TBT concentrations. While this may be generally true, since 12-week mean concentrations of seawater TBT at all our San Diego Bay sites are now < 100 ng/ℓ, the estimates are better. Even so, tissue TBT concentration can still probably only be used to detect approximate order-of-magnitude differences as originally intended in a Mussel Watch approach (Goldberg et al., 1978). This has significant implications for interpreting data from any chemical monitoring program based on measuring tissue concentrations of various contaminants for predictive purposes. Many investigators tend to over-extend the utility of tissue accumulation data and fail to discuss the pitfalls in applying these data for predictive purposes (Phillips, 1980).

Factors Modifying Threshold Responses

White (1984) has cautioned against the arbitrary use of mussel monitoring systems without developing a model to be tested. We have developed a mussel bioindicator model that emphasizes the importance of natural factors and other contaminants on mussel survival, bioaccumulation and growth (Salazar and Salazar, 1990a). Although there is a correlation between decreases in seawater TBT and increases in juvenile mussel growth rates, there are other factors that also regulate mussel growth. The extreme drop in growth rates between December 1989 and March 1990, when seawater TBT concentrations were quite low, suggests that temperature was a significant factor. The seasonal effects of temperature and nutrition on mussel growth rates are well known (Seed, 1976; Newell, 1979) and we have previously discussed their effects in detail (Salazar and Salazar, 1990a). The adverse effects of high TBT concentrations (> 100 ng/ℓ) and handling stress from weekly measurements have minimized correlations between growth rates, temperature, and chlorophyll-α in Tests 1-4, particularly at the marina sites. In Tests 5-9 growth rate was well correlated with temperature and chlorophyll-α concentrations; seasonal effects became more apparent when we switched from weekly to biweekly measurements and seawater TBT concentrations decreased.

We have previously estimated an optimum temperature for juvenile mussel growth in San Diego Bay near 20°C. This is consistent with laboratory predictions (Bayne et al., 1985). In these studies during the coldest winter test, 12-week mean temperatures at all sites were <15°C and growth rates were among the lowest we measured even though mean concentrations of seawater TBT were lower than in most other tests. During the warmest summer test, 12-week mean temperatures ranged from 20.1°C at MSF to 24.4°C at NAV. Growth rates at MSF were the highest ever measured in San Diego Bay during this particular summer test while growth rate at NAV was the lowest of all Navy sites. The highest mean temperature, 25.7°C, was measured in Test 9 at Coronado Cays, the most southern marina site in San Diego Bay. A maximum temperature of 27°C was recorded at this site. Test 9 growth rates at Coronado Cays were lower than at any other marina site except SI. There was no statistically difference in growth rates between mussels transplanted to Coronado Cays and SI. However, seawater TBT concentrations were significantly higher at SI and temperatures were significantly higher at Coronado Cays. Stress from high temperatures near 25°C have been associated with extremely adverse effects on mussels in both the laboratory and the field (Bayne et al., 1985, Wells and Gray, 1960). There are three important factors to consider in the environmental effects of seawater TBT concentration, chlorophyll-α and temperature on juvenile mussel growth. They are means, extremes and seasonal variability. The biological significance of the extreme variability has yet to be determined. Laboratory studies have shown that mussels are affected more by continuous than discontinuous exposure to copper (Davenport, 1977) and that they can detect elevated levels of copper and close to avoid exposure (Davenport and Manley, 1978). We do not know how mussels respond to order-of-magnitude shifts in seawater TBT concentrations with tidal cycle in the Shelter Island Yacht Basin previously reported (Clavell et al., 1986) or the weekly variability we measured.

Control over test conditions is the primary advantage of using laboratory toxicity tests for estimating threshold responses to contaminants. Our data emphasize the extreme variability of the nature and lead to questions of how to control and duplicate these conditions in the laboratory. Daily maximum temperature changes of up to 5°C and overall temperature ranges of over 8°C were measured. There were order-of magnitude shifts in seawater TBT and chlorophyll-α concentrations at some sites. Most laboratory studies do not include this variability or other factors that affect bioavailability of contaminants and mussel growth (Salazar, 1986). It is difficult to determine what laboratory test conditions should be used and how this relates to natural conditions. Figure 10 compares all of the growth measurements we have made in the field with those from our flow-through microcosm work (Salazar and Salazar, 1987) and lab studies in the U.K using juvenile mussels (Thain, 1986; Thain and Waldock, 1985; Thain and Waldock, 1986). There are order-of-magnitude differences in the results. There are reportedly significant differences when comparing laboratory versus field responses of mussels exposed to TBT in survival, bioaccumulation and growth (Salazar, 1989). It has been shown that maximum mussel growth in nature greatly exceed growth rates measured in the laboratory (Kiorboe et al., 1981).

Monitoring Implications

The importance in making the distinction between using bioindicators to determine the extent of contamination of TBT versus the use of bioindicators to determine environmental effects must be emphasized (Waldock et al., 1990). Chemical monitoring programs that only monitor seawater and tissue concentrations of contaminants document the extent of contamination and not environmental effects. Any assumptions of environmental effects must be inferred from other laboratory or field studies. Biological monitoring programs measure these effects directly. Field measurements provide a realistic test platform for long-term studies, but generally lack the control necessary for experimentation and establishing cause-and-effect relationships, particularly with TBT (Stebbing, 1985). Some control was gained in our experiments by transplanting mussels instead of monitoring natural populations. We were able to identify site-specific differences that could not be determined with natural populations. Pertinent questions that arise are, 1) what to measure, 2) where to measure, and 3) how often to measure it.

What to Measure

The first question to be answered is what species should be used as the biological indicator. We chose mussels because they are found in San Diego Bay, and they have been used in monitoring programs throughout the world. Other biomonitoring programs in San Diego Bay by California Mussel Watch use the Japanese Oyster *Crassostrea gigas* because they are reportedly more sensitive to TBT than mussels and the shell thickening response is reportedly specific to TBT (Stephenson et al., 1986; Smith et al., 1987; Stephenson et al., 1988). We feel this approach is flawed primarily because this species is not naturally foundin San Diego Bay nor are they cultured here. It is neither a representative endemic species or of commercial importance in San Diego Bay. An added benefit of mussels is that they can be used to evaluate a variety of contaminants. Additionally, many of the pitfalls have been thoroughly documented and accounted for (Phillips, 1980). This has not been done for shell thickening in oysters. A more generalized approach considers the environment as a whole and separates the effects attributable to TBT with frequent multiple measurements as we have done.

If it is decided to use mussels as the bioindicator of choice, where to get the mussels and what size to use to accurately assess environmental stress becomes an issue. Juveniles were used in our studies to avoid the effects of gametogenesis on growth and because we felt they were more sensitive than adults. We minimized the range of the smallest size mussel (10-12 mm) for practicable collection and to avoid the effects of gametogenesis on growth (Rodhouse et al., 1986) and minimize variability (Phillips and Segar, 1986). However, growth rate was so rapid in some tests that animals were producing gametes by the end of the test. We have also shown a transplant effect and a size effect on bioaccumulation of TBT by mussels (Salazar and Salazar, 1990a). This also has significant monitoring applications. Fischer (1983, 1988) has suggested that bioaccumulation in short-term tests with fast-growing juvenile mussels more accurately reflects recent environmental changes. An integrated monitoring program should include both juveniles and adults.

It is important to use animals from the area that is being studied. Initial mussel studies by Mussel Watch in San Diego Bay included transplants from another site in northern California (Stephenson et al., 1986). Mussel Watch frequently transplants mussels from this pristine site to identify hot-spots of contamination throughout the state by using a standard size and mussel population for a baseline. These comparisons are reasonable and informative. However, if one is attempting to make comparisons within San Diego Bay, it should be demonstrated that the surrogate population is responding like the natural endemic population. Most of the juvenile mussels died in the first San Diego Bay transplant studies by California Mussel Watch, reportedly due to transportation shock. Since these northern California mussels came from a different genetic and environmental stock, it is also possible that they are more sensitive to San Diego Bay contaminants, temperatures and other natural factors. In 67 of the 79 transplants made in our San Diego Bay studies, survival was between 89–100%. Recent electrophoretic studies have even suggested a different species of *Mytilus* in northern California (McDonald and Koehn, 1988). This potential variability must be minimized in monitoring studies.

We have attempted to detect differences between sites using growth to estimate effects and bioaccumulation to document the extent of contamination. There are apparent difficulties in using bioaccumulation in mussel tissues to distinguish differences between particular sites. This may be attributable to the way seawater TBT and tissue TBT are measured. The hydride derivatization method may not truly reflect bioavailable seawater TBT (Salazar, 1986), measured tissue TBT could be biologically inactive (Salazar and Salazar, 1990a), and wet weights of TBT in mussel tissue may obscure true tissue levels (Page and Widdows, 1990). There are unusual problems in attempting to compare wet

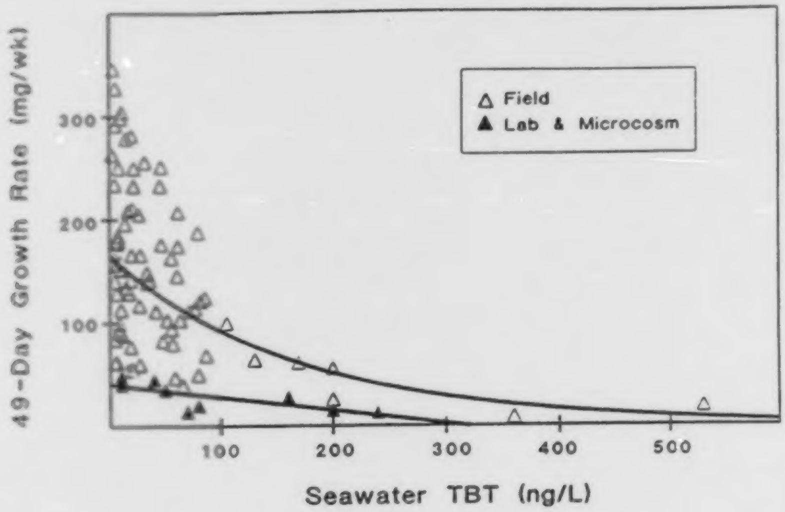

Figure 10. Comparison of 49–day growth rates for juvenile mussels transplanted in the field with juvenile mussel growth rates from labortory and microcosm studies.

weight, dry weight and radioactivity-based concentration units (Kelly et al., 1990). It may also be important to consider lipid normalized values since TBT is relatively lipophilic. However, in a recent study with DDT and PCB's, lipid normalized and non-normalized values were not much different in oysters (Sericano et al., 1990).

Where to Measure

The individual site data demonstrate the difficulty in generalizing TBT effects in an area like San Diego Bay or perhaps any estuary, and even within the Shelter Island Yacht Basin. One or two sites with extreme values can easily bias a regional mean and subsequent statistical analyses. Similar problems with extremely high values strongly biasing means in large environmental data sets have been reported from NOAA Status and Trends Programs with bivalve tissues (Uhler et al., 1989; Sericano et al., 1990). A generalized approach often de-emphasizes the variability and extreme values that could be driving biological responses. Identifying these extremes in short-term and long-term variability on a site-specific basis is essential to any monitoring program. To adequately monitor the Shelter Island Yacht Basin, all of the measurements we have made over the last 3 years would have to be concentrated in that area.

The differences in tissue levels of TBT found in our study and those reported by Valkirs et al. (1990) could be attributable to several factors. These include differences between juveniles and adults, sampling frequency, site-specific differences and perhaps even subtle differences in methodology. The natural populations sampled in their monitoring studies were generally limited to the shoreline whereas all water samples were taken in the middle of yacht basins, Navy berthing areas, etc. The primary advantage of transplants is to place animals where they would not normally settle or occur in sufficient quantities to monitor and collect water samples immediately next to the transplant. Without juvenile mussel transplants at the SID bottom site, we would not have been able to show the differences in seawater TBT concentration, tissue TBT concentration and mussel growth rates because mussels are not naturally found there.

How Often to Measure

Most monitoring programs with quarterly sampling plans, like the Navy's statutory monitoring program, are designed primarily to establish long-term trends as well as establishing an environmental baseline. A potential problem with quarterly sampling is that the biological effects may actually be occurring on a weekly scale as in our growth rate measurements or even a daily or hourly scale. Environmentally relevant data from monitoring programs must quantify temporal and spatial variability in contaminants, natural factors and biological responses (Carpenter and Huggett, 1984; Phillips and Segar, 1986). Further, this monitoring must be quantified on an equivalent time scale. Real-time or very short-term chemical and biological measurements are available. Real-time chemical sensors for heavy metals and TBT have been developed in our laboratory and have been used to document the large fluctuations with tidal cycle (Zirino et al., 1978; Clavell et al., 1986). Near real-time cellular, physiological and biochemical measures of biological stress have also been developed (Bayne et al., 1986). We used intensive sampling of seawater, tissues and mussel growth rates to quantify both the extent of TBT contamination and its environmental effects and establish short-term effects. Long-term trends were established by conducting serial seasonal transplant studies over a 3-year period. Our data could be improved significantly with the addition of real-time sampling that more accurately reflected temporal and spatial variability that might affect juvenile mussel growth.

SUMMARY

Serial seasonal transplants of juvenile mussels with frequent growth measurements and chemical sampling have documented temporal and spatial variability in TBT and its effects on growth, bioaccumulation and survival that have not previously been reported. Establishing this variability and identifying site-specific factors affecting juvenile mussel growth in San Diego Bay have facilitated environmentally relevant predictions of threshold levels for TBT effects on mussel growth. This represents the initial calibration of the mussel bioindicator for TBT effects and establishes a significant refinement in the use of mussels as biological indicators. Based on the data presented here, questions have been raised regarding the environmental significance of chemical and biological monitoring programs that may be applicable to all bioindicator responses and monitoring programs. Crucial questions regarding environmental fate and effects of TBT remain unanswered. Sampling must document temporal and spatial variability of the contaminant in question, other contaminants, natural factors that affect biological responses (particularly the bioindicator being used) as well as temporal and spatial variability in the natural biological response. This generic issue of defining an environmentally relevant monitoring program is much more important than the specific issue of TBT effects. All monitoring programs should include chemical measurements to document the extent of contamination and biological indicators to estimate environmental effects.

ACKNOWLEDGEMENTS

We thank Team Hydride (S. Frank, J. Guerrero, M. Kram, M. Stallard, P. Stang, A. Valkirs and J. Vaughn) for measuring TBT in over 700 water samples. A. Valkirs has helped review every TBT paper and experimental design. We also thank S. Cola and L. Kear for measuring TBT in more than 100 tissue samples and always making publication deadlines on short notice. A special acknowledgement is extended to B. Davidson for helping collect, sort and transplant over 2, 000 mussels as well as managing, manipulating and testing these extremely large data bases since we began this work in 1987. Biological monitoring was supported by the Office of Chief of Naval Research and chemical monitoring by the Naval Sea Systems Command.

REFERENCES

Abel, R., N. J. King, J. L. Vosser and T. G. Wilkinson. 1986. The control of organotin use in antifouling paint - the U.K.'s basis for action. In: Proceedings, Oceans 1986 Conference, Washington D. C., 23-25 Sept 1986, Organotin Symposium, Vol. 4, pp. 1314-1323.

Abel, R., R. A. Hathaway, N. J. King, J. L. Vosser and T. G. Wilkinson. 1987. Assessment and regulatory actions for TBT in the U.K. In: Proceedings, Oceans 1987 Conference, Halifax, Nova Scotia, Canada, 28 Sept - 1 Oct 1987, Organotin Symposium. Vol. 4 pp. 1314-1319.

Alzieu, C. 1986. TBT detrimental effects on oyster culture in France – Evolution since antifouling paint regulation. In: Proceedings, Oceans 1986 Conference, Washington, D. C., 23-25 Sept 1986, Organotin Symposium, Vol. 4, pp. 1130-1134.

Bayne, B. L., D. A. Brown, K. Burns, D. R. Dixon, A. Ivanovici, D. R. Livingstone, D.M. Lowe, N. M. Moore, A. R. D. Stebbing, and J. Widdows. 1985. The effects of stress and pollution on marine animals. Praeger Special Studies, Praeger Scientific, New York. 384 pages.

Cairns, J., Jr. 1988. What constitutes field validation of predictions based on laboratory evidence: In: W. J. Adams, G. A. Chapman and W. G. Landis (Eds.), Aquatic Toxicology and Hazard Assessment: Tenth Volume, ASTM STP 971. American Society for Testing and Materials, Philadelphia, pp. 361-368.

Cairns, J., Jr. and A. L. Buikema, Jr. 1984. Verifying predictions of environmental safety and harm. In: H. H. White (Ed.), Concepts in Marine Pollution Measurements, Maryland Sea Grant Publication, University of Maryland, College Park. pp. 81-111.

Carpenter, J. H. and R. J. Huggett. 1984. Meaningful chemical measurements in the marine environment – transition metals. In: H. H. White (Ed.), Concepts in Marine Pollution Measurements. Maryland Sea Grant Publication, University of Maryland, College Park, pp. 379-403.

Champ, M. A. and F. L. Lowenstein. 1987. TBT: The dilemma of high-technology antifouling paints. Oceanus 30(3):69-77.

Chapman, G. A. 1983. Do organisms in laboratory toxicity tests respond like organisms in nature? In: W. E. Bishop, R. D. Cardwell and B. B. Heidolph (Eds.), Aquatic Toxicology and Hazard Assessment: Sixth Symposium, ASTM STP 802, American Society for Testing and Materials, Philadelphia, pp. 315-327.

Chapman, P. M. and E. R. Long. 1983. The use of bioassays as part of a comprehensive approach to marine pollution assessment. Mar. Poll. Bull. 14(3):81-84.

Clavell, C., P. F. Seligman, P. M. Stang. 1986. Automated analysis of organotin compounds: a method for monitoring butyltins in the marine environment. In: Proceedings, Oceans 1986 Conference, Washington D. C., 23-25 Sept 1986, Organotin Symposium, Vol. 4, pp. 1152-1154.

Davenport, J. 1977. A study of the effects of copper applied continuously and discontinuously to specimens of *Mytilus edulis* (L.) exposed to steady and fluctuating salinity levels. J. Mar. Biol. Assoc. U. K. 57:63-74.

Davenport, J. and A. Manley. 1978. The detection of heightened sea-water copper concentrations by the mussel *Mytilus edulis*. J. Mar. Biol. Assoc. U. K. 58:843-850.

Farrington, J. W., E. D. Goldberg, R. W. Risebrough, J. H. Martin and V. T. Bowen. 1983. "Mussel Watch" 1976-78: an overview of the trace-metal, DDE, PCB, hydrocarbon, and artificial radionuclide data. Environ. Sci. Tech. 17:490-496.

Fischer, H. 1983. Shell weight as an independent variable in relation to cadmium content of molluscs. Mar. Ecol. Prog. Ser. 12:59-75.

Fischer, H. 1988. *Mytilus edulis* as a quantitative indicator of dissolved cadmium. Final study and synthesis. Mar. Ecol. Prog. Ser. 48(2):163-174.

Goldberg, E. D., V. T. Bowen, J. W. Farrington, G. Harvey, J. H. Martin, P. L. Parker, R. W. Risebrough, W. Robertson, E. Schneider, E. Gamble. 1978. The mussel watch. Environ. Conserv. 5(2):101-125.

Goldberg, E. D., M. Koide, V. Hodge, A. R. Flegal and J. Martin. 1983. U.S. Mussel Watch: 1977-1978 results on tracemetals and radionuclides. Estuar. Coast. Shelf Sci. 16:69-93.

Hall, L. W. and S. J. Bushong. 1990. A review of acute effects of tributyltin compounds on aquatic biota. In: M. A. Champ and P. F. Seligman (Eds.), Tributyltin: Environmental Fate and Effects. In Prep. Elsevier.

Kelly, J. R., D. T. Rudnick, R. D. Morton, L. A. Buttel and S. N. Levine. 1990. Tributyltin and invertebrates of a seagrass ecosystem: exposure and response of different species. Mar. Environ. Res. 29:245-276.

Key, D. R. S. Nunny, P. E. Davidson, and M. A. Leonard. 1976. Abnormal shell growth in the Pacific oyster (*Crassostrea gigas*): Some preliminary results from experiments undertaken in 1975. Inter. Council. Explor. Sea CM 1976/K:11.

Kiorboe, T., F. Mohlenberg and 0. Nohr. 1981. Effect of suspended bottom material on growth and energetics in *Mytilus edulis*. Mar. Biol. 61:283-288.

Kram, M. L., P. M. Stang and P. F. Seligman. 1989. Adsorption and desorption of tributyltin in sediments of San Diego Bay and Pearl Harbor. Appl. Organomet. Chem. 3:523-536.

Malins, D. C., H. 0. Hodgins, U. Varanasi, W. D. MacLeod, Jr., B. B. McCain and S.-L. Chan. 1984. Chemical measurements of organic pollutants and effects criteria. In: H. H. White (Ed.), Concepts in Marine Pollution Measurements, Maryland Sea Grant Publication, University of Maryland, College Park, pp. 405-426.

Martin, M. 1985. State mussel watch: Toxics surveillance in California. Mar. Poll. Bull. 16(4):140-146.

McDonald, J. H. and R. K Koehn. 1988. The mussels *Mytilus galloprovincialis* and *M. trossulus* on the Pacific coast of North America. Mar. Biol. 99:100-118.

Moller, H. 1987. The marine ecologist – scientist or advocate of nature? Mar. Poll. Bull. 18(6):267-270.

Newell, R. C. 1979. Biology of intertidal animals. 3rd ed., Marine Ecological Surveys Ltd., Faversham, Kent, 781 pp.

Page, D. S. and J. Widdows. 1990. Temporal and spatial variation in levels of alkyltins in mussel tissues: A toxicological interpretation of field data. In press: Mar. Envir. Res.

Parrish, P. R., K. L. Dickson, J. L. Hamelink, R. A. Kimerle, K. J. Macek, F. L. Mayer, Jr. and D. I. Mount. 1988. Aquatic toxicology: Ten years in review and a look at the future. In: W. J. Adams, G. A. Chapman and W. G. Landis (Eds.), Aquatic Toxicology and Hazard Assessment: 10th Volume, ASTM STP 971, American Society for Testing and Materials, Philadelphia, pp. 7-25.

Peddicord, R. K 1984. What is the meaning of bioaccumulation as a measure of marine pollution effects? In: H. H. White (Ed.), Concepts in Marine Pollution Measurements, Maryland Sea Grant Publication, University of Maryland, College Park, pp. 249-259.

Phillips, D. J. H. 1980. Qualitative Aquatic Biological Indicators. Applied Science Publishers, Ltd., London. 488 pp.

Phillips, D. J. H. and D. A. Segar. 1986. Use of bio-indicators in monitoring conservative contaminants: Programme design imperatives. Mar. Poll. Bull. 17(1):10-17.

Rodhouse, P. G., J. H. McDonald, R. I. E. Newell and R. K Koehn. 1986. Gamete production, somatic growth and multiple-locus enzyme heterozygosity in *Mytilus edulis*. Mar. Biol. 90:209-214.

Salazar, M. H. 1986. Environmental significance and interpretation of organotin bioassays. In: Proceedings, Oceans 1986 Conference, Washington D. C., 23-25 Sept 1986, Organotin Symposium, Vol. 4, pp. 1240-1245.

Salazar, M. H. 1989. Mortality, growth and bioaccumulation in mussels exposed to TBT: differences between the laboratory and the field. In: Proceedings, Oceans 1989 Conference, 18-21 September 1989, Seattle, Washington, Organotin Symposium, Vol. 4, pp. 1188-1197.

Salazar, M. H. and M. A. Champ. 1988. Tributyltin and water quality: A question of environmental significance. In: Proceedings, Oceans 1988 Conference, Baltimore, Maryland, 31 Oct - 2 Nov, 1988, Organotin Symposium. Vol. 4, pp. 1497-1506.

Salazar, M. H. and S. M. Salazar. 1985. Ecological evaluation of organotin-contaminated sediment. Naval Ocean Systems Center Technical Report 1050.

Salazar, M. H. and S. M. Salazar. 1987. TBT effects on juvenile mussel growth. In: Proceedings, Oceans 1987 Conference, Halifax, Nova Scotia, Canada, 28 Sept - 1 Oct 1987, Organotin Symposium. Vol. 4 pp. 1504-1510.

Salazar, M. H. and S. M. Salazar. 1988. Tributyltin and mussel growth in San Diego Bay. In: Proceedings, Oceans 1988 Conference, Baltimore, Maryland, 31 Oct - 2 Nov, 1988, Organotin Symposium. Vol. 4, pp. 1188-1195.

Salazar, M. H. and S. M. Salazar. 1990a. Mussel field studies: Mortality, Growth and Bioaccumulation. In: M. A. Champ and P. F. Seligman (Eds.), Tributyltin: Environmental Fate and Effects. In Prep. Elsevier.

Salazar, M. H. and S. M. Salazar. 1990b. Mussels as bioindicators: the utility of growth and bioaccumulation to assess site-specific differences in TBT contamination and its effects. Mar. Envir. Res. In press.

Salazar, S. M., B. M. Davidson, M. H. Salazar, P. M. Stang and K. Meyers-Schulte. 1987. Field assessment of a new site-specific bioassay system. In: Proceedings, Oceans 1987 Conference, Halifax, Nova Scotia, Canada, 28 Sept - 1 Oct 1987, Organotin symposium, Vol. 4 pp. 1461-1470.

Seed, R. 1976. Ecology. Marine mussels: Their ecology and physiology, pp 13-65. Edited by B. L. Bayne. Cambridge: Cambridge University Press.

Seligman, P. F., J. G. Grovhoug and K E. Richter. 1986. Measurement of butyltins in San Diego Bay, Ca. In: Proceedings, Oceans 1986 Conference, Washington D. C., 23-25 Sept 1986, Organotin Symposium, Vol. 4, pp. 1289-1296.

Seligman, P. F., J. G. Grovhoug, R. L. Fransham, B. Davidson and A. 0. Valkirs. 1990. U.S. Navy statutory monitoring of tributyltin in selected U.S. harbors. Naval Ocean Systems Center Technical Report 1346, June 1990.

Sericano, J. L., E. L. Atlas, T. L. Wade and J. M. Brooks. 1990. NOAA's status and trends mussel watch program: chlorinated pesticides and PCB's in oysters (*Crassostrea virginica*) and sediments from the Gulf of Mexico, 1986-1987. Mar. Environ. Res. 29: 161-203.

Short, J. W. and J. L. Sharp. 1989. Tributyltin in bay mussels (*Mytilus edulis*) of the Pacific coast of the United States. Environ. Sci. Tech. 23(6):740-743.

Smith, D. R., M. D. Stephenson, J. Goetzl, G. Ichikawa and M. Martin. 1987. The use of transplanted juvenile oysters to monitor the toxic effects of tributyltin in California waters. In: Proceedings, Oceans 1987 Conference, Halifax, Nova Scotia, Canada, 28 Sept - 1 Oct 1987, Organotin Symposium. Vol. 4, pp. 1511-1516.

Stang, P. M., D. R. Bower and P. F. Seligman. 1989. Stratification and tributyltin variability in San Diego Bay. J. Appl. Organomet. Chem. 3:105-114.

State of California. 1988. Title 3, California Code of Regulations. Section 6488-6489.

Stebbing, A. R. D. 1985. Organotins and water quality – some lessons to be learned. Mar. Poll. Bull. 16(10) 383-390.

Stephenson, M. D., D. R. Smith, J. Goetzl, G. Ichikawa and M. Martin. 1986. Growth abnormalities in mussels and oysters from areas with high levels of tributyltin in San Diego Bay. In: Proceedings, Oceans 1986 Conference, Washington D. C., 23-25 Sept 1986, Organotin Symposium, Vol. 4, pp. 1246-1251.

Stephenson, M., L. Aronson, J. Goetzl, G. Ichikawa, K. Paulson, D. Moore, M. Martin, H. Lenihan and J. Oliver. 1988. Report on TBT in San Diego Harbor. California Dept. Fish and Game, Moss Landing Marine Laboratories, Submitted to The California State Regional Water Quality Control Board.

Stromgren, T. and T. Bongard. 1987. The effect of tributyltin oxide on growth of *Mytilus edulis*. Mar. Poll. Bull. 18(1):30-31.

Thain, J. E. 1986. Toxicity of TBT to bivalves: Effects on reproduction, growth and survival. In: Proceedings, Oceans 1986 Conference, Washington, D. C., 23-25 Sept 1986, Organotin Symposium, Vol. 4, pp. 1306-1313.

Thain, J. E. and M. J. Waldock. 1985. The growth of bivalve spat exposed to organotin leachates from antifouling paints. ICES Paper CM 1985/E:28, International Council for the Exploration of the Sea, Copenhagen.

Thain, J. E. and M. J. Waldock. 1986. The impact of tributyltin (TBT) antifouling paints on molluscan fisheries. Wat. Sci. Tech. 18:193-202.

Uhler, A. D., T. H. Coogan, K. S. Davis, G. S. Durell, W. G. Steinhauer, S. Y. Freitas and P. D. Boehm. 1989. Findings of tributyltin, dibutyltin and monobutyltin in bivalves from selected U.S. coastal waters. Envir. Toxic. Chem. 8:971-979.

U.S. Environmental Protection Agency. 1986. Initiation of a special review of certain pesticide products containing tributyltins used as antifoulants; availability of support document. Federal Register 51:778.

U.S. Environmental Protection Agency. 1987. Ambient aquatic life water quality advisories for tributyltin. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratories, Duluth, Minnesota, Narragansett, Rhode Island.

U.S. Department of Navy. 1985. Fleet Wide Implementation of Organotin Antifouling Hull Paints. Federal Register 50:25748.

U.S. Public Law. 1988. Public law 100-330 102 Stat. 605: Organotin Antifouling Paint Control Act of 1988. 16 June 1988.

Valkirs, A. 0., P. F. Seligman, and R. F. Lee. 1986. Butyltin partitioning in marine waters and sediments. In: Proceedings, Oceans 1986 Conference, Washington D.

C., 23-25 Sept 1986, Organotin Symposium, Vol. 4, pp. 1165-1170.

Valkirs, A. O., B. M. Davidson, and P. F. Seligman. 1987. Sublethal growth effects and mortality to marine bivalves from long-term exposure to tributyltin. Chemosphere 16(1):201-220.

Valkirs, A. O., B. Davidson, R. L. Fransham, J. G. Grovhoug and P. F. Seligman. 1990. Long term monitoring of tributyltin in San Diego Bay, California. In: M. A. Champ and P. F. Seligman (Eds.), Tributyltin: Environmental Fate and Effects. In Prep.

Wade, T. L., B. Garcia-Romero and J. M. Brooks. 1988. Tributyltin contamination in bivalves from United States coastal estuaries Environ. Sci. Tech. 22(12):1488-1493.

Waldock, M. J. 1986. TBT in U.K. estuaries, 1982-86. Evaluation of the environmental problem. In: Proceedings, Oceans 1986 Conference, Washington, D. C., 23-25 Sept 1986, Organotin Symposium, Vol. 4, pp. 1324-1330.

Waldock, M. J., J. E. Thain and M. E. Waite. 1990. An assessment of the value of shell thickening in *Crassostrea gigas* as an indicator of exposure to tributyltin. In: M. A. Champ and P. F. Seligman (Eds.), Tributyltin: Environmental Fate and Effects. In Prep.

Wells, H. W. and I. E. Gray. 1960. The seasonal occurrence of *Mytilus edulis* on the Carolina coast as a result of transport around Cape Hatteras. Biol. Bull. 119:550-559.

White, H. H. 1984. Mussel madness: use and misuse of biological monitors of marine pollution. In: H. H. White (Ed.), Concepts in Marine Pollution Measurements. Maryland Sea Grant Publication, University of Maryland, College Park, pp. 325-337.

White, H. H. and M. A. Champ. 1983. The great bioassay hoax, and alternatives. In: R. A. Conway and W. P. Gulledge (Eds.), Hazardous and Industrial Solid Waste Testing: Second Symposium. ASTM STP 805. American Society for Testing and Materials. pp. 299-312.

Zirino, A., S. H. Lieberman and C. Clavell. 1978. Measurement of Cu and Zn in San Diego Bay by automated anodic stripping voltammetry. Environ. Sci. Tech. 12(1):73-80.

Zuolian, C. and A. Jensen. 1989. Accumulation of organic and inorganic tin in blue mussel, *Mytilus edulis*, under natural conditions. Mar. Poll. Bull. 20(6):281-286.

AN APPLICATION OF "REAL-TIME" MONITORING IN DECISION MAKING: THE NEW BEDFORD HARBOR PILOT DREDGING PROJECT

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ABSTRACT

A decision-making framework was established for assessing the impacts of a pilot dredging study at the New Bedford Harbor, CA, Superfund site. Concern over possible environmental impacts due to dredging at this site necessitated that a monitoring program be implemented to ensure that unacceptable water quality impacts did not occur during this project. Consequently, criteria were derived, a management committee assembled, and a "real-time" monitoring plan designed. Because many existing chemical concentrations in the water column and indigenous biota exceeded Federal and state water quality limits, site-specific chemical and biological criteria were established. A committee of environmental managers from Federal and state government was established with the authority to assess and modify the operation on a daily basis. Finally, a "real-time" monitoring plan was implemented in which water samples were collected, analyzed within 16 hours, and the data supplied to the management committee in order to assess the environmental impact of the previous days' operation. The combined use of site-specific criteria and a "real-time" decision making management process allowed for successful completion of this project with a minimal effect on water quality.

EXTENDED SUMMARY

New Bedford Harbor (NBH) is located along Buzzards Bay between the cities of New Bedford and Fairhaven, MA. Since the 1940's, electronics and manufacturing companies in the area have discharged effluents containing polychlorinated biphenyls (PCBs) into the harbor. High PCB concentrations in harbor sediments were first documented in 1974 (Connelly and St. John, 1988), with PCB concentrations as high as 100,000 parts per million (ppm) in some areas of the upper harbor. In 1982, the site was added to the Environmental Protection Agency's (EPA) National Priorities List of hazardous waste sites slated for cleanup under the Superfund Act.

A feasibility study conducted by EPA in 1984 proposed several alternatives for the remediation of NBH. One option common to most remediation alternatives was the dredging of contaminated sediments. Federal, state, and local officials, as well as the public, expressed concern that the resuspension of sediments during dredging may cause the release of contaminants that would affect biota at more distant areas in the harbor and Buzzards Bay. Others cited potential pollution problems from contaminated water (leachate) leaking from the proposed disposal site (Averett and Francigues, 1988).

In order to address these concerns, EPA Region I, in conjunction with the U.S. Army Corps of Engineers (COE), initiated the NBH Pilot Dredging Project to establish the impacts of various dredging and disposal options on a small scale with relatively low (with respect to NBH) contaminated sediments (PCB concentrations approximately 100 ppm). Information derived from this project would be used to determine the most environmentally safe methods for use in a possible large scale remediation of the most contaminated areas of the NBH Superfund Site.

The overall goal of the Pilot Project was to determine the feasibility of various dredging and disposal options for removing and sequestering highly contaminated sediments in NBH. This included assessing whether or not it was practical from an engineering perspective, as well as determining if the operations could be completed without causing unacceptable environmental impacts. The engineering aspect of the project assessed three shallow-water dredges capable of removing sediment with minimal resuspension. In addition, two disposal methods were evaluated: 1) a confined disposal facility (CDF), which required construction of a containment dike partially in-water and partially on land; and 2) a confined aquatic disposal cell (CAD), an in situ underwater disposal method (Otis, 1987). The results of these engineering operations are reported elsewhere (Otis, 1989).

A second objective implicit in the overall goal was to determine whether the engineering operations could be completed in such a manner as not to cause unacceptable damage to the environment. The decision-making process used to assess the environmental acceptability of this project is the topic of this summary.

Because of the high PCB concentrations in the sediments to be dredged during the Pilot Project (100 ppm), it was necessary to make rapid assessments as to the environmental "acceptability" of the operations. The evaluation of possible unacceptable contamination of the water column due to dredging during the Pilot Project was complicated by the fact that Federal and state water quality standards for PCBs (U.S. EPA, 1980) and certain heavy metals (U.S. EPA, 1985) were exceeded in NBH under preoperational baseline conditions. In addition, the U.S. Food and Drug Administration (FDA) action level for PCBs in seafood in NBH was exceeded (Kolek and Ceurvels, 1981).

These special conditions necessitated the development of a distinctive site-specific monitoring/management strategy for the Pilot Project. This framework included several unique aspects: 1) development of a set of site-specific Decision Criteria for assessing water and tissue chemical concentrations and biological effects, 2) establishment of a panel of environmental managers, Decision Criteria Committee, to use those data in a timely manner, and 3) design and implementation of a monitoring program to provide the necessary environmental data to the Committee in a rapid timeframe (12-24 hours). This approach provided an effective feedback loop to evaluate, modify or terminate the dredging operation if the Decision Criteria were exceeded.

Each aspect of this strategy was successfully implemented. First, site-specific Decision Criteria were established at two strategic locations within the harbor. The philosophy adopted for establishing criteria values was that short-term, near-field elevations in contaminant concentrations or biological effects would be evaluated against long-term improvements in water quality, provided that no far-field effects were observed. Using this rationale, criteria were established for a number of physical, chemical, and biological parameters based on data collected prior to the initiation of dredging (Nelson and Hansen, in press).

Secondly, a Decision Criteria Committee was formed with representatives from each of the major participants involved in the study: EPA Region I, the COE, the Massachusetts Department of Environmental Protection, and EPA's Environmental Research Laboratory, Narragansett, RI (ERL–N). This committee was empowered to make decisions on a daily basis if there were impacts attributable to the operation. Possible corrective actions to limit adverse effects due to the project ranged from altering operational procedures to temporarily halting the operation or termination of the study.

Finally, a monitoring plan was developed and implemented by ERL-N to collect samples during the operational phases of the project, complete sample analysis within 24 hours, and transmit the resultant information to the Committee for comparison with the Decision Criteria values.

The chemical and biological monitoring data indicated that the dredging operation had a minimal effect on existing water quality; the only criterion exceeded was PCB water concentration. On the four occasions when elevated PCB concentrations were detected, they were attributed to a specific causative operational procedure or meteorological event. Operational modifications were implemented effectively, thus limiting elevations in water column PCB concentrations.

It may be unrealistic to expect to complete a Superfund remediation at an aquatic site with absolutely zero shortterm impact. However, this program successfully established a set of limits (Decision Criteria) beyond which the impact was considered unacceptable, and a mechanism (real-time monitoring program) which provided the information necessary for environmental managers (Decision Criteria Committee) to effectively oversee this project to completion.

REFERENCES

Averett, D. E. and N. R. Francingues, Jr. 1988. A case study: dredging as a remedial action alternative for New Bedford Harbor, Massachusetts. Superfund '88. Proceedings of the 9th National Conference. The Hazardous Materials Control Research Institute, pp 338-342.

Connelly, J. P. and J. P. St. John. 1988. Application of a mathematical food chain model to evaluate remedial alternatives for PCB-contaminated sediments in New Bedford Harbor, Massachusetts. Superfund '88. Proceedings of the 9th National Conference. The Hazardous Materials Control Research Institute, pp 359-362.

Folek, A. and R. Ceurvels. 1981. Polychlorinated biphenyl (PCB) analysis of marine organisms in the New Bedford area, 1976-1980. 29 p. in Commonwealth of Massachusetts, Division of Marine Fisheries. Publication No. 12265-36-1001-81-CR.

Nelson, W. G. and D. J. Hansen. In press. Development and use of site-specific chemical and biological criteria for assessing the New Bedford Harbor Pilot Dredging Project. Environmental Management.

Otis, M. J. 1987. Pilot study of dredging and dredged material disposal alternatives. Draft. U.S. Army Corps of Engineers, New England Division, Waltham, MA.

Otis, M. J. 1989. New Bedford Harbor Superfund Pilot Study: Evaluation of dredging and dredged material disposal. Interim Report. U.S. Army Corps of Engineers, New England Division, Waltham MA.

U.S. EPA. 1980. Ambient Water Quality Criteria for Polychlorinated Biphenyls. EPA 440/5-80-068, 197 pp.

U.S. EPA. 1985. Ambient Water Quality Criteria for Copper – 1984. EPA 440/5-84-031, 142 pp.

EPA'S ENVIRONMENTAL MONITORING AND ASSESSMENT PROGRAM: AN ECOLOGICAL STATUS AND TRENDS PROGRAM

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ABSTRACT

The U.S. Environmental Protection Agency has initiated the Environmental Monitoring and Assessment Program (EMAP) to monitor the status and trends of the nation's near coastal waters, forests, wetlands, agroecosystems, surface waters, and arid lands. The program is also intended to evaluate the effectiveness of Agency policies at protecting ecological resources occurring in these systems. Monitoring data collected for all these natural resources will be integrated for national status and trends assessments. The near coastal component of EMAP consists of estuaries, coastal waters, coastal and estuarine wetlands, and Great Lakes. The country's near coastal resources have been regionalized and classified, an integrated sampling strategy has been developed, and quality assurance/quality control procedures and data management designs have been implemented. EPA and NOAA have agreed to coordinate and, to the extent possible, integrate near coastal component of EMAP with NOAA National Status and Trends Program. A demonstration project was jointly conducted in the estuaries of the mid-Atlantic states (Chesapeake Bay to Cape Cod) in the summer of 1990. In 1991, monitoring will continue in the mid-Atlantic estuaries and will be initiated in estuaries in part of the Gulf of Mexico.

INTRODUCTION

Environmental regulatory programs have been estimated to cost more than $70 billion annually, yet the means to assess their effect on the environment over the long term do not exist. While regulatory programs are based upon our best understanding of the environment at the time of their development, it is critical that long-term monitoring programs be in place to confirm the effectiveness of these programs in achieving the environmental goals and to corroborate the science upon which they are based.

The U.S. Environmental Protection Agency (EPA), the U.S. Congress, and private environmental organizations have long recognized the need to improve our ability to document the condition of our environment. Congressional hearings in 1984 on the Monitoring Improvement act concluded that, despite considerable expenditures on monitoring, federal agencies could assess neither the status of ecological resources nor the overall progress toward legally-mandated goals of mitigating or preventing adverse ecological effects. In the last decade, articles and editorials in professional journals of the environmental sciences have repeatedly called for the collection of more relevant and comparable ecological data and easy access to those data for the research community. The most commonly suggested tools for accomplishing these goals include a national ecological survey and a bureau of environmental statistics.

Affirming the existence of a major gap in our environmental data and recognizing the broad base of support for better environmental monitoring, the EPA Science Advisory Board recommended that EPA initiate a program that would both monitor ecological status and trends, as well as to develop innovative methods for anticipating emerging problems before they reach crisis proportions (SAB 1988). EPA was encouraged to become more active in ecological monitoring because its regulatory responsibilities require quantitative, scientific assessments of the complex effects of anthropogenic activities on ecosystems. The Environmental Monitoring and Assessment Program (EMAP) is EPA's response to all of these recommendations.

EPA's Office of Research and Development, in concert with several other federal agencies, is developing EMAP to monitor the condition of ecological resources to ensure that our environmental protection efforts are achieving the desired results. When fully implemented, EMAP will be able to respond to the following questions:

- What proportion of the nation's ecological resources are degrading or improving, where, and at what rate?
- What are the likely causes of the observed degraded conditions?
- What is the current status, extent, and geographic distribution of our ecological resources?
- Are control and mitigation programs effective in maintaining or improving the quality of our resources?

OBJECTIVES OF EMAP

To provide the information necessary to address the above questions and the goal of the program, EMAP has established the three following objectives:

1. To estimate the current status, extent, changes, and trends in indicators of the condition of the nation's ecological resources on a regional basis with known confidence,
2. To monitor indicators of pollutant exposure and habitat condition and seek associations between man-induced stresses and ecological conditions, and
3. To provide periodic statistical summaries and interpretative reports on ecological status and trends to the EPA Administrator and the public.

APPROACH AND RATIONALE

Assessing the status and trends for the nation's ecological resources requires data collected in a standardized manner, over large geographic scales, for long periods of time. Such assessments cannot be accomplished by aggregating data from the many individual, short-term monitoring programs that have been conducted in the past and are being conducted currently. Differences in the parameters measured, the collection methods used, timing of sample collection, and program objectives severely limit the value of historical monitoring data for conducting regional assessments (Wolfe et al. 1987; NRC 1989, 1990a, 1990b).

EMAP proposes to monitor a defined set of parameters (i.e., indicators of environmental quality) on a regional scale, over a period of decades, using standardized sampling methods with a probability-based sampling design. These characteristics distinguish EMAP from other monitoring programs and provide the data for preparing the regional and national scale assessment that are needed to address the environmental issues of the 1990s and beyond (Reilly 1989; Thomas 1988a, 1988b).

Local programs that measure the same parameters and sample in a manner compatible with EMAP will be able to use EMAP products to obtain a regional and national perspective with which to evaluate the seriousness of local problems. This will assist them in two ways: (1) by determining whether local problems are unique and, (2) by facilitating detection of problems that are more easily measured on regional or national scales (e.g., determination of whether apparent declines in values resources are associated with regional changes in climate or is more likely attributable to regional or local changes in pollutant loadings.)

NEAR COASTAL COMPONENT OF EMAP

The near coastal component of EMAP has established its inland boundary as the limit of tidal influence. The outer boundary is the continental shelf break. Ecosystems occurring between these boundaries that ultimately will be sampled are the following:

Estuarine and coastal wetlands – submerged lands characterized by periodic or constant saturation and the presence of vegetation adapted to or tolerant of saturated soils.

Estuaries – semi-enclosed bodies of water that have a free connection with the open sea and an inflow of freshwater that mixes with the seawater.

Coastal waters – waters lying over the continental shelf.

Great Lakes – large freshwater bodies not affected by marine currents; each lake has a unique, complex circulation pattern.

At the present time, EMAP does not have the financial resources to implement regional monitoring programs in all near coastal ecosystems simultaneously. Therefore, a phased implementation is proposed that focuses much of the initial efforts on estuaries (USEPA 1990).

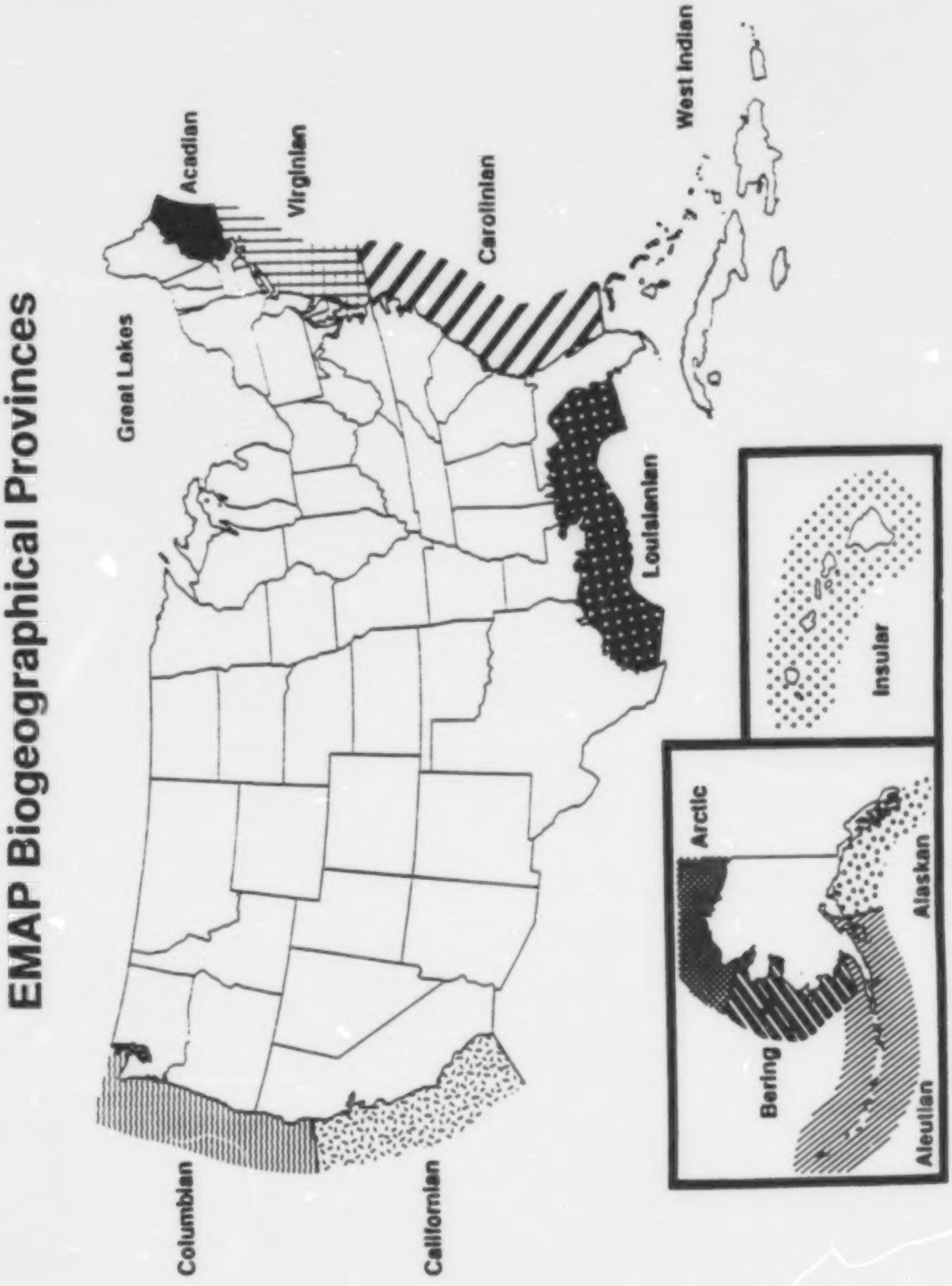
EMAP Biogeographical Provinces
Great Lakes
Acadian
Virginian
Carolinian
West Indian
Louisianian
Columbian
Californian
Insular
Arctic
Bering
Alaskan
Aleutian

FIGURE 1. EMAP–Near Coastal regionalization scheme based on biogeographical provinces.

SAMPLING DESIGN

The sampling design for the near coastal component of EMAP has three elements:

- A regionalization scheme for partitioning near coastal resources into regions with similar ecological properties that constitute reasonable reporting units.
- A classification scheme to define subpopulations of interest (e.g., classes of estuaries, types of wetlands) that can be sampled using a common approach.
- A statistical design that will obtain unbiased estimates of the status and trends of near coastal ecological resources cost effectively.

The regionalisation scheme is used to divide the nation's near coastal resources into a series of biogeographical provinces (Figure 1). Initially, the field activities will be implemented in the Virginian Province, with other provinces added in subsequent years. By 1995, all provinces in the continental U.S. should be included in the sampling program.

A classification scheme is used to subdivide estuaries within a province into classes that have similar physical features and are likely to respond to stressors in a similar manner. The classes that are defined include (1) large, continuously distributed estuaries (e.g., Chesapeake Bay, Long Island Sound), (2) large tidal rivers (e.g., Potomac, Delaware Rivers), and (3) small, discretely distributed estuaries, bays, inlets, and tidal creeks and rivers (e.g., Barnegat Bay, Elizabeth River). The purposes of classifying estuaries into categories having similar attributes (e.g., size, shape, resource distributions) are (1) a common sampling design can be applied to each class, (2) the variability in condition within a class should be less than which occurs among classes, reducing the number of samples necessary to characterize a class accurately, and (3) the degree of confidence with which inferences can be made about systems within a class that are not sampled is increased.

A critical issue that must be addressed is how best to represent the ecological condition of near coastal environments with limited financial resources and relatively few samples. It is obvious that one or two samples, from one or two locations, at one time of the day, in a specific season of a particular year cannot characterize the ecological condition of even a small estuary. Such a sampling program is justified only if it can be demonstrated that parameters that are indicative of the overall ecological condition can be identified and a population approach to sampling can be used to characterize resources. That is, resources and locations that are sampled can be used to make inferences about unsampled resources and locations. One of the goals of 1990 EMAP–Near Coastal field effort is to make this demonstration.

EMAP-Near Coastal does not have the resources to characterize natural variability or to assess status in all seasons. Therefore, sampling will be limited to a confined portion of the year (i.e., an index period), when measured parameters are expected to show the greatest response to pollution stress and within-season variability is expected to be reduced. EMAP–Near Coastal has selected summer as the appropriate index period. For most near coastal ecosystems in the northern hemisphere, mid-summer (July–August) is a period when dissolved oxygen concentrations are most likely to approach stressful low values (Holland et al. 1977; Officer et al. 1984), and the cycling and adverse effects of contaminant exposure are maximal because of low dilution flows and high temperatures (Connell and Miller 1984; Sprague 1985). In addition, fauna and flora are usually abundant during summer, increasing the probability of collecting the organisms required to complete assessments.

Within each estuarine class, elements of systematic, random, and fixed location sampling are used. Large, continuously distributed estuaries are sampled using a randomly placed systematic grid. Grid points are about 18 km apart, and the entire estuary is sampled. Large tidal rivers are sampled along systematically spaced lateral transects. Transects are located about 25 km apart. The starting point for the first transect at the mouth of the river (between river kilometre 0–25) are randomly selected. Two sampling points are located on each transect; one is randomly selected and one is an index sample. The goal of the index sample is to use scientific judgement to identify sampling locations that can be used to determine if degraded conditions occur in a system without having to conduct intensive surveys. The index sample site will be located in a depositional, muddy environment where sediments are accumulating, and the potential for exposure to low dissolved oxygen concentration and/or to contaminants is high. Small, relatively discrete estuaries will be sampled using a population approach. First, a list of all small estuaries is defined and placed in order according to latitude. Then the estuaries are classified into groups of four and one estuary from each group is randomly selected for sampling without replacement. Two sampling points are located in each small estuary that is sampled: one is randomly selected and one is an index sample. Regional scale information from index sites will be combined with similar information from randomly selected locations.

Index samples will be used to estimate the proportion of sampling sites in small estuaries and tidal river segments that have unacceptable (or acceptable) indicator values in places that are particularly vulnerable to pollution impacts. However, the index samples are biased and cannot be used alone to estimate the extent of degradation. When regional scale information from index sites is combined with similar information from randomly selected locations, robust statements can be made about the proportion of systems that have pollution problems in highly vulnerable sites as well as about the extent and magnitude (i.e., areal extent) of degradation for the population of small estuaries and tidal river segments.

When fully implemented, EMAP-Near Coastal will operate on a four-year sampling cycle, with approximately one fourth of the total number of samples needed to make an overall assessment collected in each year. Regional interpretative assessments will be prepared every four years by combining the data collected over the four-year cycle. Such a multi-year baseline reduces the confounding effect of year-related phenomena (e.g., weather) to the assessment process. Multi-year baseline are particularly important for establishing the effectiveness of management actions (USEPA 1983a, 1983b). Annual assessments can be made with the data collected during that year; however, these annual assessments will have a higher degree of uncertainty than assessments based on the full four-year sampling program.

INDICATORS OF ENVIRONMENTAL QUALITY

Many studies have defined the major problems facing the nation's estuaries and near coastal waters (e.g., OTA 1987; USEPA 1983a; NRC 1989; NOAA 1988). In general, these studies conclude that the major environmental issues for near coastal ecosystems are those that adversely affect the maintenance of balanced indigenous populations of fish, shellfish, and other biota including the following:

- Increases in the amount of water that has low dissolved oxygen concentration levels;
- Eutrophication;
- Chemical and microbial contamination of water, sediments, and biological tissue;
- Habitat modification; and
- Cumulative impacts of more than one of the above.

The EMAP–Near Coastal indicator strategy was developed to address these problems and their associated impacts on valued ecosystem attributes.

EMAP-Near Coastal does not have the resources to monitor all ecological parameters of concern to the public, Congress, scientists, and decision-makers. Therefore, a defined set of parameters that serve as indicators of environmental quality will be measured. Indicators will be selected to be:

- Related to ecological condition in a way that can be quantified and interpreted;
- Applicable across a range of habitats and biogeographic provinces;
- Valued by and of concern to society; and
- Quantifiable in a standardized manner with a high degree of repeatability.

The selection of indicators that will be used by EMAP-Near Coastal is an ongoing process. It is anticipated that a number of years will be required to develop a complete list of indicators. The selection process consists of the following steps:

- Identification of values ecosystem attributes and stressors that affect them;
- Development of a conceptual source–receptor model that links valued ecosystem attributes to stressors;
- Using the conceptual model to identify candidate indicators;
- Evaluation and classification of candidate indicators into categories (core, development, research) using evaluation criteria that are generic to all EMAP resources (e.g., forest, arid land, agroecosystems);
- Testing and evaluation of indicators to assess their ability to discriminate between polluted and unpolluted sites;
- Conducting regional scale demonstration projects to show the feasibility of indicators and the value of indicator data to characterizing overall ecosystem status; and
- Periodic re-evaluation of indicators.

Categories of indicators that will be identified and sampled by EMAP–Near Coastal include the following:

Response indicators – Measurements that quantify the integrated response of ecological resources to individual or multiple stressors. Examples include measures of the condition of individuals (e.g., frequency of tumours or other pathological disorders in fish), populations (e.g., abundance, biomass), and communities (e.g., species composition, diversity).

Exposure indicators – Physical, chemical, and biological measurements that quantify pollutant exposure, habitat degradation, or other causes of degraded ecological condition. Examples include contaminant concentrations in the water, sediment, and biological tissues; the acute toxicity of sediments to endemic or sensitive biota and dissolved oxygen concentrations.

Habitat indicators – Physical, chemical, and biological measurements that provide basic information about environmental setting. Examples include water depth, salinity, sediment characteristics, and temperature. Habitat indicators will be used to normalize values for exposure and response indicators across environmental gradients. Habitat indicators will also be used as a basis for defining subpopulations of interest for assessments.

Stressor indicators – Economic, social, or engineering measures that can be used to identify sources of pollution and poor ecological condition. Examples include human demographics, land use patterns, discharge records from sewage treatment facilities, freshwater inflows, and pesticide usage on the watershed. Stressor data will be gathered primarily from existing federal and state programs (e.g., NOAA's National Coastal Pollution Discharge Inventory (NCPDI), wetland acreage and extent from USFWS National Wetland Inventory), from other EMAP task groups (e.g., extent and distribution of forests, atmospheric deposition of pollutants), and from local permitting/planning agencies. EMAP-Near Coastal recognizes that it also will have to spend some of its resources on measuring stressors.

The relationships among indicator categories are summarized in Figure 2. Information on exposure, habitat, and stressor indicators will be used to identify potential factors contributing to the status and trends of response indicators. A list of indicators that were used in the first year of the program is provided in Table 1. In the first year, EMAP-Near Coastal is oversampling indicators and use the data collected to develop a reduced list of indicators that can be applied to characterize overall estuarine condition accurately when the program is fully implemented. The over-sampling is necessary because indicators of estuarine condition that are acceptable to the public and scientists and have been demonstrated to be appropriate to apply at regional scales are not well developed.

ANALYSIS AND INTEGRATION

Integration and synthesis of EMAP-Near Coastal data into assessments of the condition of estuaries is a formidable challenge. Assessment results must be scientifically defensible and presented in a manner that can be understood by non-technical audiences. Unfortunately, estuarine science has not developed measures of environmental condition of estuaries that are accepted by scientists and understood by the public and other non-technical audiences.

To accomplish its objectives, EMAP-Near Coastal will conduct the following types of analyses:

- Status assessments;
- Trends evaluations; and
- Diagnostic evaluations including identification of factors that may be affecting status and trends.

The analysis approach for status assessments will be hierarchial. First the overall condition of estuarine resources will be quantified using response indicators to define the extent and magnitude of pollution problems. Then, this integrated assessment will be decomposed to define associations between exposure, habitat, and stressor indicators and to identify likely causes and relative contributions of various stresses to problems.

A principal graphical representation of EMAP status information will be cumulative distribution functions (CDFs). CDFs were chosen because essential information on both central tendency (e.g., mean, median) and extreme values can be summarized in an easily interpreted graphical format (Figure 3). CDFs will be prepared for response indicators for each estuary class, for all estuaries within the region, and eventually for all estuaries nationally. CDFs also will be prepared for selected exposure indicators and to characterize habitat conditions using habitat indicators.

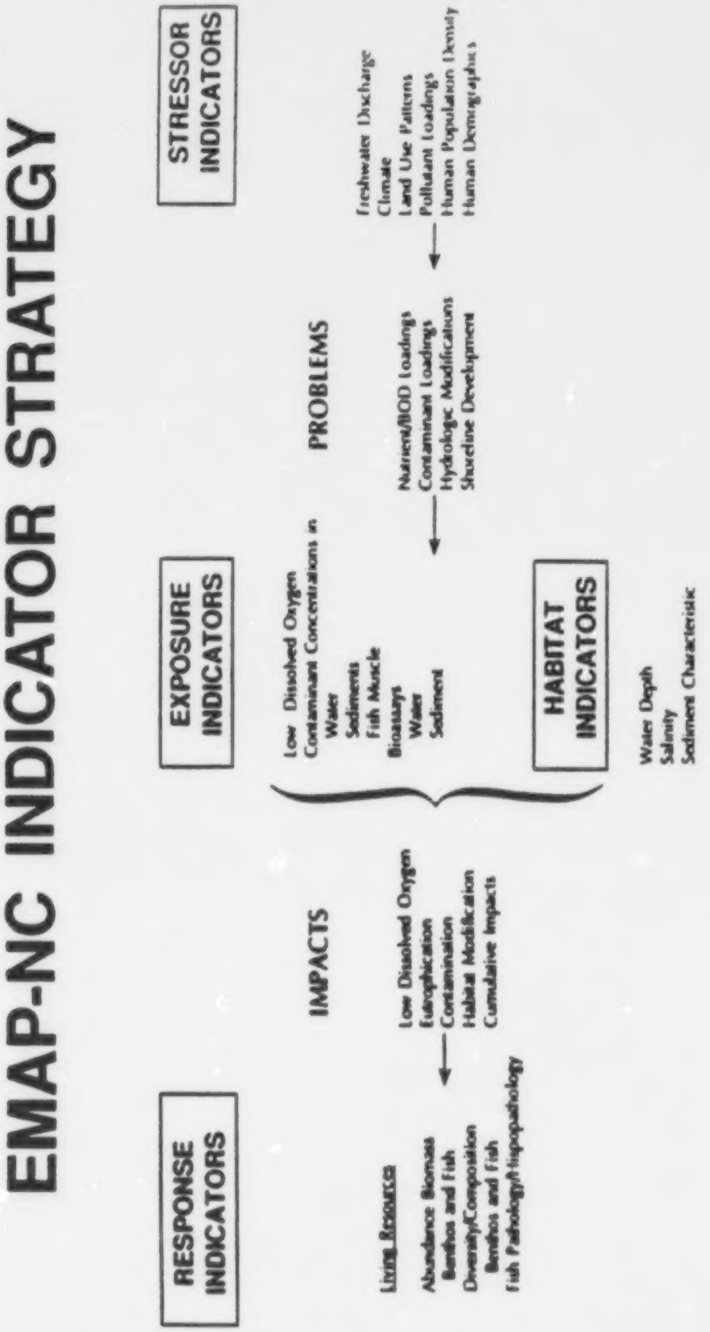
EMAP-NC INDICATOR STRATEGY
RESPONSE INDICATORS
Living Resources
Abundance Biomass
Benthos and Fish
Diversity/Composition
Benthos and Fish
Fish Pathology/Histopathology
IMPACTS
Low Dissolved Oxygen
Eutrophication
Contamination
Habitat Modification
Cumulative Impacts
EXPOSURE INDICATORS
Low Dissolved Oxygen
Contaminant Concentrations in
Water
Sediments
Fish Muscle
Bioassays
Water
Sediment
HABITAT INDICATORS
Water Depth
Salinity
Sediment Characteristic
PROBLEMS
Nutrient/BOD Loadings
Contaminant Loadings
Hydrologic Modifications
Shoreline Development
STRESSOR INDICATORS
Freshwater Discharge
Climate
Land Use Patterns
Pollutant Loadings
Human Population Density
Human Demographics

FIGURE 2. Overview of indicator strategy for EMAP–Near Coastal.

TABLE 1. List of EMAP–Near Coastal indicators by major category. The manner in which indicators are related to the major environmental problems and impacts is also shown.

CATEGORY	PROPOSED INDICATORS
Response	Benthic species composition and biomass Gross pathology of fish Dissolved oxygen concentration Fish community composition Relative abundance of large burrowing shellfish Histopathology of fish Apparent RPD depth
Exposure	Sediment contaminant concentration Sediment toxicity Contaminants in fish flesh Contaminants in large bivalves Water column toxicity Continuous DO measurements Contaminant screening
Habitat	Salinity Sediment characteristics Water depth
Stressor	Fresh water discharge Climatic fluctuations Pollutant loadings by major category Land use patterns of watershed by major categories Human population density/demographics

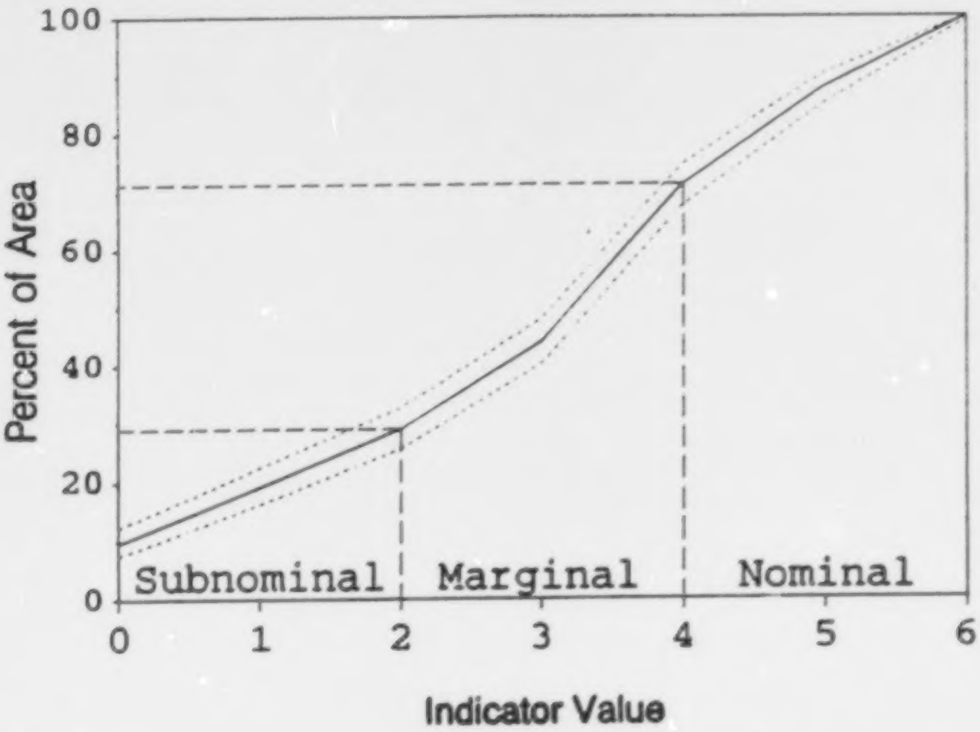

FIGURE 3. Example cumulative distribution function (CDF). Dotted lines represent confidence limits.

The approach to trend assessment will consist of sampling a portion (e.g., one-fourth) of the sampling sites each year in a manner that ensures geographic dispersion and repeating the cycle on a regular basis (e.g., every 4 years). Annual estimates of status can be evaluated individually or aggregated with other years to establish multi-year baselines that are more stable than annual estimates. Multi–year baselines are particularly useful for measuring trends and for evaluating the effectiveness of pollution control programs.

Although individual response indicators are important measures of specific aspects of environmental condition, the goal of EMAP–Near Coastal is to provide answers to questions with an holistic perspective of estuarine systems. Multiple statements (i.e., multiple CDFs) about the status and trends of the nation's estuaries, each based on a different response indicator, present information that may confuse many EMAP clients. Single, integrated statements about the overall status of estuarine resources are more easily communicated and understood. Therefore, EMAP-Near Coastal must develop an Estuarine Condition Index (ECI) that integrates the data collected for multiple response indicators into a single CDF describing the status of estuarine resources.

DATA MANAGEMENT

EMAP-Near Coastal will use a distributed data management system. In this system, data are produced at a number of remote locations, where samples are processed. Results are then transferred to a central site where they are verified to be reasonable and are integrated into the Near Coastal Information Management System (NCIMS). The NCIMS will include data in both raw and summary form to minimize costly redundant analysis (NRC 1990a). Information on study characteristics, institutional and organizational structures, sampling methods, sample status, data format, quality assurance, key scientists involved in the generation of each data set, and data access support will be available for all data sets. Data management reports that summarize the types, volume, and quality of data, as well as a list of specific data sets that are available, will be prepared and distributed to potential users frequently (i.e., approximately every two years). For additional information on EMAP-Near Coastal data management, readers should refer to the EMAP-Near Coastal Data Management System Plan (Rosen et al. 1990).

QUALITY ASSURANCE

EMAP-Near Coastal will employ EPA's data quality objective (DQO) approach to ensure that the type, amount, and quality of data collected are adequate to meet program goals and that analysis results have quantifiable and acceptable levels of uncertainty. The DQO process is an iterative approach, balancing costs against uncertainty, to achieve a desired or acceptable level of data quality (Figure 4). The first step in the DQO process consists of determining the level of uncertainty that the decision makers, who will use the data, are willing to accept. Then, the uncertainty associated with the measurement program is estimated. The two estimates are compared and the sampling program modified (e.g., intensity of sampling increased or decreased, sampling methods altered) until the proper balance between costs and uncertainty is achieved. Once an acceptable level of uncertainty has been established, quality control and quality assessment procedures are applied to each program element (e.g., field sampling, laboratory analysis, transfer of information to a data base, and data analysis) to ensure that the specified level of quality is attained and maintained.

Because regional data with which to estimate spatial and temporal variability within the summer index period are either unavailable or unaccessible for most, if not all, of the proposed indicators, it will not be possible for EMAP-Near Coastal to implement DQOs during the first year or two of the program. Accordingly, the first year's program will be implemented using measurement quality objectives (MQOs). MQOs establish the acceptable level of uncertainty for field and laboratory methods. MQOs differ from DQOs in that they do not consider spatial and temporal variability in estimating uncertainty levels. The DQO uncertainty level for each indicator is based on the available scientific literature for sampling, processing, and measurement methods or a manufacturer's specification for a given instrument (Plumb 1981; Home and McIntyre 1984; SeaBird Electronics, Inc. 1987; Pollard et al. 1990).

The data collected using MQOs during the first several years of EMAP-Near Coastal will be used to measure the uncertainty associated with the regional measurement program for each indicator. This information will then be evaluated, acceptable DQOs defined, and the sampling program modified as necessary to address program objectives. For additional information on the EMAP-Near Coastal quality assurance program, readers should refer to the Quality Assurance Project Plan developed for the 1990 Demonstration Project in estuaries of the Virginian Province (Pollard et al. 1990). EMAP-Near Coastal also has developed field collection and laboratory processing methods manuals that standardize sampling and processing operations for this study (Strobel 1990; Graves 1990).

1990 DEMONSTRATION PROJECT

As a first step in accomplishing the objectives of EMAP-Near Coastal, a Demonstration Project was implemented in the estuaries of the Virginian Province in 1990. The major goals of this Demonstration Project are to evaluate the utility of the EMAP-sampling design and approach and, at the same time, collect information necessary to develop a technically sound and cost effective sampling program that can implemented over the long term. The specific goals of the 1990 Virginian Province Demonstration Project include the following:

- Demonstrate the value of regional monitoring data collected in a standard way, measuring a defined set of parameters, using a robust sampling design as a basis for status assessment;
- Identify, test, and evaluate indicators of environmental quality for estuaries that can be applied over broad regions;
- Develop standardized sampling and processing methods for evaluating estuarine environmental quality;
- Evaluate alternative sampling designs and approaches for establishing a regional and national monitoring network in estuaries;
- Develop analysis procedures for converting monitoring data into information useful for the public, Congress, environmental decision makers, policy analysts, and the scientific community; and
- Identify and resolve logistical problems associated with conducting a regional/national scale monitoring program in estuaries.

EMAP-Near Coastal is being implemented in the Virginian Province because there is a general public perception that estuaries in this area are rapidly deteriorating. Additionally, many of the estuaries in this region have been intensively investigated by scientists, and a considerable amount of information has been available for use in designing the Demonstration Project. Finally, many management decisions for this region are forthcoming, including development of a restoration plan for the New York Harbour complex and development and evaluation of management plans of many large estuaries, including Delaware Bay, Chesapeake Bay, and Long Island Sound. Development of these plans presents an opportunity to demonstrate how EMAP information can assist in the formulation of environmental programs and policies.

The 1990 Virginian Province Demonstration Project includes a number of program enhancements. These special studies include:

- An indicator testing and evaluation program that will evaluate the ability of indicators to discriminate between polluted and unpolluted environments;
- Temporal sampling for some indicators (e.g., dissolved oxygen concentration) extending beyond the boundaries of the anticipated index period to better define starting and ending times for the index period;
- Repeated measurements of selected indicators (e.g., dissolved oxygen concentration, fish community characteristics, and contaminants in fish flesh) during the index period to assess their stability and suitability for application in the sampling design;and
- Intensive spatial sampling conducted at a subset of stations (i.e., Delaware River, Delaware Bay, Indian River Estuary) to evaluate the advantages and disadvantages of sampling at alterative spatial scales.

The reporting associated with the 1990 Virginian Province Demonstration Project includes the following:

- A Near Coastal Program Plan for 1990: Estuaries (USEPA 1990), which describes the detailed program plans for the Demonstration Project;
- An Example Assessment Report (Frithsen 1990), that presents examples of the kinds of assessment information that EMAP-Near Coastal will produce;
- A Demonstration Project Activities Summary, due in the winter 1990, that summarizes the data collected, describes the status of data records, identifies and discusses problems and issues

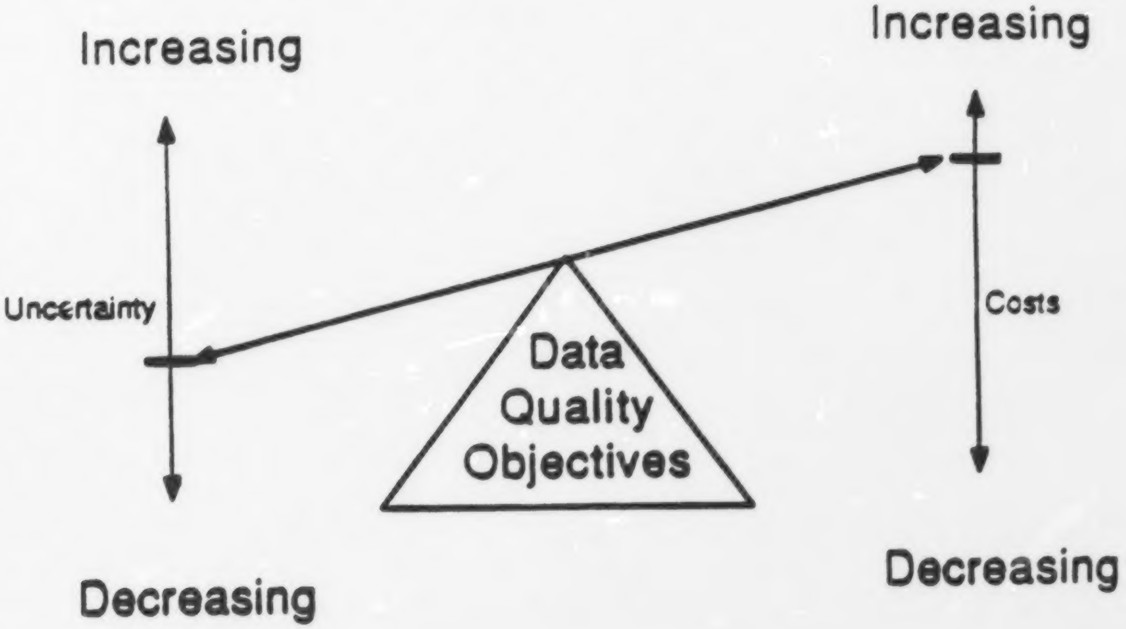
Increasing
Uncertainty
Decreasing
Data Quality Objectives
Increasing
Costs
Decreasing

FIGURE 4. Role of data quality objectives in obtaining balance between available resources and level of uncertainty.

encountered during the field program, and develops recommendations for improving logistical activities during the implementation phase;

- A Demonstration Project Assessment Report, due in the summer 1991, that makes a status assessment for the Virginian Province based on one year of data and presents the findings of the indicator testing and evaluation program, intensive spatial sampling efforts, and the evaluation of alternative sampling designs. The report will provide the technical basis for the design of future EMAP-Near Coastal monitoring efforts in the Virginian Province and elsewhere.

COORDINATION

Meeting the objectives of EMAP requires close coordination among many offices within EPA and with other federal, state, and local agencies involved in monitoring activities. Although EMAP is funded by the Office of Research and Development, other offices within EPA (e.g., Office of Marine and Estuarine Protection) have participated actively in its development. Near Coastal has coordinated with each of the National Estuary Programs in the Virginian Province about activities in 1990 and beyond. Coordination will occur with other National Estuary Programs and ongoing EPA programs (e.g., Gulf of Mexico Program) before monitoring activities are implemented in these regions.

Both NOAA and EPA have mandates to conduct a broad range of research and monitoring activities to assess the effects of pollution on coastal and estuarine environments. There are similarities and differences between existing NOAA and EPA programs; however, the combined results of both agencies' programs serve the national interest more than the results of individual programs. It is the intention of NOAA and EPA to cooperate and coordinate, to the greatest extent possible, to integrate estuarine and coastal monitoring, research, and assessment activities and to ensure that data collected by EMAP-Near Coastal and NOAA National Status and Trends Program augment and complement each other to the maximum extent possible.

The framework for cooperating and coordinating monitoring and research activities between NOAA and EPA is the NOAA/EPA Committee for Coastal and Estuarine Environmental Quality Monitoring. This committee was created to ensure coordination and exchange of information between the two agencies on coastal monitoring, research, and assessment. The joint committee has held monthly meetings since October 1989. The purpose of these meetings has been to exchange planning information and to identify opportunities for joint complementary activities. As a result of the activities of this committee, a joint NOAA/EPA quality assurance program has been implemented for sampling near coastal environments. Through the joint committee, NOAA has assisted EPA in the development and evaluation of coastal and estuarine environmental quality indicators by participating in workshops, providing data for retrospective analyses, and reviewing EPA plans and analysis results. The joint NOAA/EPA committee recently developed and executed a Memorandum of Understanding (MOU) that defines continued interagency cooperation and interaction and provides a framework for integrating activities of NOAA National Status and Trends Program and EMAP-Near Coastal into a unified national monitoring and assessment program for estuarine and coastal waters (see appendix B in USEPA 1990).

Coordination with NOAA and other federal agencies, as well as with other offices within EPA, avoids duplicative monitoring efforts and allows existing data to be used to maximum benefit. This coordination should lead to the incorporation of historical baselines established by other agencies, such as NOAA's baselines on contaminant concentrations in sediments and bivalves, into EMAP-Near Coastal analyses. It will also lead to the incorporation of EMAP-Near Coastal data into the analyses and assessments accomplished by other agencies. The regional-scale assessments resulting from EMAP-Near Coastal, in combination with the ongoing characterization work of NOAA, will provide a substantial portion of the technical information needed to (1) characterize existing conditions and define coastal environmental problems; (2) coordinate the design and implementation of regional monitoring and assessment activities; and (3) identify, assess, and recommend management strategies and solutions that will enhance and protect regional coastal environmental quality.

FUTURE FIELD ACTIVITY PLANS

In 1991, estuarine sampling will continue in the Virginian Province, and a Demonstration Project will begin in the estuaries of the Louisianian Province. Progress reports on these activities will be available in 1992. During the summer of 1992, EMAP-Near Coastal expects to begin sampling estuaries in the Carolinian Province.

ACKNOWLEDGEMENTS

Although this work was supported by the U.S. Environmental Protection Agency, it has not been subjected to Agency review and therefore does not necessarily reflect the views of the Agency; no official endorsement should be inferred. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

REFERENCES

Connell, D.W., and G.J. Miller. 1984. Chemistry and Ecotoxicology of Pollution. New in San Francisco Bay: An overview. In: Estuarine Process, Vol. 2, 48–62. M.L. Wiley, ed. New York: Academic Press.

Frithsen, J., M. Fabrizio, A.F. Holland, and G. Sauls. 1990. EMAP-Near Coastal Illustrative Example Assessment Report. U.S. Environmental Protection Agency. Office of Research and Development, Narragansett, Rhode Island.

Graves, R.L. 1990. Draft Environmental Monitoring and Assessment Program Near Coastal Demonstration Project Laboratory Methods Manual. U.S. Environmental Protection Agency, Office of Research and Development.

Holland, A.F., N.K. Mountford, and J.A. Mihursky. 1977. Temporal variation in upper bay mesohaline benthic communities:
I. The 9–m mud habitat. *Chesapeake Sci.* 18:370–378.

Holme, N.A. and A.D. McIntyre. 1984. Methods for Study of Marine Benthos. London: Blackwell Scientific Publications.

National Oceanic and Atmospheric Administration (NOAA) 1988. Federal plan for ocean pollution research development, and monitoring: Fiscal years 1988-1992. Prepared by the National Ocean Pollution Program Office for the National Ocean Pollution Policy Board. Rockville, MD.

National Research Council (NRC). 1989. Contaminated Marine Sediments-Assessment and Remediation. Washington, DC: National Academy Press.

NRC. 1990a. Managing Troubled Waters: The Role of Marine Environmental Monitoring. Washington, DC: National Academy Press.

NRC. 1990b. Monitoring Southern California Coastal Waters. National Academy Press.Washington, DC.

Office of Technology Assessment (OTA). 1987. Wastes in Marine Environments. Washington, DC.

Officer, C.B., R.B. Biggs, J.L. Taft, L.E. Cornin, M.A. Tyler, and W.R. Boynton. 1984. Chesapeake Bay anoxia: Origin, development, and significance. *Science* 223:22–27.

Plumb, R.H. 1981. Procedures for handling and chemical analyses of sediment and water samples. Prepared for the U.S. Environmental Protection Agency/Corps of Engineers Technical Committee on Criteria for Dredge and Fill Material. Published by Environmental Laboratory, U.S. Waterways Experiment Station, Vicksburg, Mississippi. Technical Report EPA/CE-81-1.

Pollard, J.E., K.M. Perez, T.C. Chaing, R. Valente, C. Strobel, and J. Rosen. 1990. Environmental monitoring and assessment program near coastal demonstration project quality assurance project plan. Environmental Monitoring Systems Laboratory, U.S. EPA, Cincinnati, OH.

Reilly, W.K. 1989. Measuring for environmental results, *EPA Journal* 15(3):2–4.

Rosen, J.S., J. Beaulieu, H. Hughes, H. Buffum, J. Copeland, R. Valente, J.F. Paul, A.F. Holland, S.C. Schimmel, C. Strobel, J.K. Summers, J.K. Scott, and J. Parker. 1990. Environmental Monitoring and Assessment Program Data Management System for Near Coastal Demonstration Project. U. S. Environmental Protection Agency, Office of Research and Development, Narragansett, Rhode Island.

Science Advisory Board (SAB). 1988. Future risk: Research strategies for the 1990s. The Report of the Research Strategies Committee. Science Advisory Board, EPA to Lee M. Thomas, Administrator, EPA.

SeaBird Electronics, Inc. 1987. Sealogger CTD. SeaBird Electronics, Inc. Bellevue, WA.

Sprague, J.B. 1985. Factors that modify toxicity. In: Fundamentals of Aquatic Toxicology. G.M. Rand and S.R. Petrocelli, ed. New York: Hemisphere Publication Corp.

Strobel, C.J. 1990. Environmental Monitoring and Assessment Program Near Coastal Component 1990 Demonstration Project Field Operations Manual. U.S. Environmental Protection Agency. Office of Research and Development.

Thomas, L.M. 1988a. The E in Environmental Protection Agency. *Environmental Protection Agency Journal* 14 (1): 3-4.

Thomas, L.M. 1988b. Environmental Regulation Challenges We Face. *Environmental Protection Agency Journal* 14(2):2–3.

U.S. Environmental Protection Agency (USEPA). 1983a. Chesapeake Bay: A profile of environmental change. Prepared for the U.S. Congress by the Environmental Protection Agency, Region 3, Philadelphia, PA.

USEPA. 1983b. Technical support manual: Waterbody surveys and assessments for conducting use attainability analyses. Vol. II: Estuarine systems. Office of Water and Standards.

USEPA. 1990. Environmental Monitoring and Assessment Program Near Coastal Program Plan for 1990: Estuaries. EPA 600/4–90/033. U.S. Environmental Protection Agency, Office of Research and Development, Narragansett, Rhode Island.

Wolfe, D.A., M.A. Champ, D.A. Flemer, and A.J. Mearns. 1987. Long-term biological data sets: Their role in research, monitoring, and management of estuarine and coastal marine systems. Estuaries 10:181–193.

EFFLUENT AND AMBIENT TOXICITY PROGRAMS IN THE SAN FRANCISCO BAY REGION

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ABSTRACT

The Regional Water Quality Control Board is conducting an evaluation of toxicity in discharges and receiving waters of the San Francisco Bay region. The ongoing Effluent Toxicity Characterization Program, sponsored by 20 major dischargers, is providing data on species sensitivity, effluent variability and toxicity test precision. The Ambient Toxicity Program has provided data on the spatial and temporal distribution of toxicity at numerous locations near the margins and in the open waters of the estuary. In some cases, effluent toxicity studies were linked to ambient toxicity surveys to assess their value in predicting the potential for receiving water toxicity.

The results of these ongoing programs have demonstrated that toxicity tests are a useful tool in assessing risks of toxicity, even within a complex estuary. Inter-laboratory comparisons indicate that test precision of chronic tests is equivalent to precision of chemical analyses. Furthermore, concurrent effluent and ambient testing have shown that risk assessments based on effluent studies combined with dilution assessments are reasonably predictive of ambient toxicity.

The information gained from both programs is being used to determine the need for Toxicity Reduction Evaluations, to derive effluent limits, and to refine the Region's Toxicity Control Program.

INTRODUCTION

The 1986 Water Quality Control Plan for the San Francisco Basin includes a Toxicity Control Program for evaluating and controlling toxicity associated with point and non-point discharges, including municipal and industrial effluents, dredged sediment disposal, urban runoff, and agricultural drainage. The Basin Plan is developed by the San Francisco Bay Regional Water Quality Control Board (Regional Board), a State agency that regulates water quality of San Francisco Bay and its tributaries within a nine county jurisdiction. The Regional Board receives its legislative authority from two sources: the federal Clean Water Act (through an agreement with the Environmental Protection Agency), and from the State of California's Porter-Cologne Act (Water Code).

Two components of the Toxicity Control Program are the Effluent Toxicity Characterization Program, which evaluates toxicity of municipal and industrial effluents, and the Ambient Toxicity Program, which surveys toxicity in San Francisco Bay waters. These programs are designed to provide data to reduce uncertainties in assessing risks to the estuary due to toxicity.

The Effluent Toxicity Characterization Program is providing data on No Observed Effect Concentrations (NOEC's) of permitted discharges, using both acute and chronic (critical life stage) toxicity tests. NOEC's are then compared with Instream Waste Concentrations (IWC's), to determine the level of risk. We consider the risk of toxicity in the receiving water as significant when the NOEC < IWC. Toxicity Reduction Evaluations are required when the number and magnitude of significantly toxic test reaches specified levels.

The Ambient Toxicity Program is providing data on toxicity in Bay waters. To date, we have surveyed five bay marshes, twelve open water ("background") locations, one dredging event, and a slough. Four of the five surveyed marshes receive secondary treated wastewater from permitted dischargers, as well as non-point inputs. The fifth marsh is a National Wildlife Refuge and receives pollutants only from non-point sources. Two of the marsh surveys were conducted simultaneously with effluent toxicity studies and dilution assessments to determine the extent to which ambient toxicity could be predicted by effluent toxicity.

METHODS

Effluent Toxicity Characterization Program

To date, twenty major (>20 million gallons per day) municipal and industrial dischargers are participating in the Effluent Toxicity Characterization Program. The Program Guidelines (Anderson et al., 1987), adopted by the Regional Board in 1987, fully describe the various phases of the program, which include a quality assurance test round and two data generation phases, the Screening and Variability Phases. The dischargers within the first group of participants are currently in the Variability Phase of the program. A second group of dischargers will commence the program in July,1991.

A quality assurance/quality control (QA/QC) test round, conducted in August, 1988, was used to determine the ability of commercial and discharger-operated toxicity testing laboratories to conduct the tests required by the program. Laboratories passing the Quality Assurance round were placed on a list of eligible laboratories for participation in the Program. To obtain Regional Board eligibility, laboratories were required to conduct three chronic tests (fathead minnow, *Ceriodaphnia* and either the larval mollusc or sea urchin fertilization tests) simultaneously, using blind toxicants supplied by the Regional Board. Laboratories were required to submit test data within twenty-one days. Eligibility was determined by test results falling within known ranges for the selected toxicants and by the quality of documentation.

QA/QC activities during later phases of the program included distribution of a technical questionnaire soliciting comments on the chronic test protocols and site visits to each of the participating toxicity testing laboratories. Responses to the questionnaire were evaluated and discussed at a follow-up meeting. A technical memorandum, setting forth recommendations and additional protocol requirements was then distributed to all program participants. Site visits included review of facilities and raw data sheets for toxicity tests conducted under this program.

The goal of the Screening Phase was to determine the three most sensitive tests for a particular effluent. Each effluent was screened for toxicity using six acute and five chronic tests, conducted simultaneously. At least one fish, one invertebrate and one plant were required for both the acute and chronic batteries. In addition, a mix of fresh, brackish and marine species was required for screening. The NOEC's generated by the Screening Phase testing were used to determine the three most sensitive tests. These three tests comprised the test battery used for repeated testing during the Variability Phase.

The Variability Phase of the Program was designed to evaluate effluent variability, as measured with toxicity tests. Each discharger is conducting up to eighteen tests over the period of a year, using a battery of the three most sensitive tests, determined from Screening Phase data. The scheduling of the tests is based on an evaluation of existing information on effluent quality, including routine and non-routine monitoring of levels of conventional pollutants, previous toxicity test data, and information on the frequency and potential effects of treatment process shut-downs and maintenance operations.

Study plans for both the Screening and Variability Phases must be submitted by the dischargers for approval by the Regional Board before commencing work. A Screening Phase report and two Variability Phase reports (six month progress report and final report) are also required. These reports summarize toxicity test results, water quality data, and test statistics. In addition, the reports must include raw data (laboratory bench sheets) and computer print-outs of statistical test runs. Toxicity test data must also be submitted in a standardized format on a computerized disk.

Data from the QA/QC round were evaluated to determine laboratory preparedness and inter-laboratory test precision. Data from the screening phase were analyzed to determine if there were trends in the distribution and frequency of acute and chronic toxicity and to assess the usefulness of requiring a broad spectrum of tests in initial screening. The six month progress reports provided data for interim risk assessments. NOEC values were compared with IWC's for each discharge. For this assessment, our Region defines the IWD as the effluent concentration at the edge of the Zone of Initial Dilution. Discharges receiving less than 10:1 initial dilution received no dilution credits, so the IWC was considered equal to 100% effluent. A Toxicity Reduction Evaluation trigger was established as six significantly toxic tests (NOEC<IWC) or three significantly toxic tests, if the NOEC was less than half the IWC in all three tests.

When final Variability Phase Reports are received, Regional Board staff will evaluate data to develop further recommendations for Toxicity Reduction Evaluations, as well as permit limits and program revisions.

Ambient Toxicity Program

Methods used in conducting ambient toxicity surveys are full, described in Anderson et al. (1990b). As a brief summary, ambient toxicity surveys were conducted once or twice in each of five marshes during 1989. Two of the marshes, Hayward Marsh and Mountain View Marsh, were created from secondary treated wastewater which flows into San Francisco Bay or a tributary to the Bay, after passing through several basins. In contrast, the marshes surveyed near San Jose/Santa Clara and Sunnyvale are natural marshes that receive advanced treated sewage (secondary plus nitrification and filtration). The San Francisco Bay National Wildlife Refuse is a marsh system that includes several sloughs and a creek, which receive some urban runoff.

Surveys of twelve Bay "background" sites, located in deeper, relatively wellmixed areas, were conducted four times between March, 1989 and April, 1990. In addition, one survey was conducted in the vicinity of a steel refinery (USS Posco) discharge and another survey was conducted during a dredging operation.

Surveys of the two natural (San Jose/Santa Clara and Sunnyvale) marshes and USS Posco were conducted at low slack tide to characterize "worst-case" conditions. These three surveys were linked to studies of effluent toxicity, conducted by the dischargers' contract laboratories. In addition, the sewage treatment plants at San Jose/Santa Clara and Sunnyvale conducted concurrent dye studies to determine dilution within the survey area.

Surveys of the reclaimed marshes were not linked to tidal flow since tidal dilution is non-existent (Mountain View) or minimal (Hayward).

Survey design varied among locations. The number of sample stations in the marsh, USS Posco and dredging surveys ranged from six to ten. Twelve stations were sampled during each or the four Bay background surveys. Water from each of the sampling stations was tested using three to five species. Sampling methods also varied, depending on field conditions. Subsurface water was sampled by boat, from shore, or from bridges or piers. Precautions were taken to minimize contamination. Salinity adjustments of the sampled waters were necessary in many cases. These were performed with natural seawater brine, commercial salt formulations and mineral waters.

RESULTS

Effluent Toxicity Characterization Program

Ten laboratories participated in the QA/QC round. Data indicated that the marine and freshwater short-term chronic tests are generally reproducible and relatively precise, despite varying animal stocks, dilution waters and laboratory operating conditions (Anderson and Norberg, 1990).

The coefficients of variation (CV's) for the *Ceriodaphnia* reproduction and survival test and the fathead minnow growth and survival tests were 29% and 31%, respectively, based on results of all ten laboratories. Six laboratories conducted the echinoderm fertilization and four conducted the mollusc (using oysters) development tests. The CV's for these tests were 16.6% and 37%, respectively.

All ten laboratories passed the QA/QC round, and thus were eligible to participate in the program. This indicated that laboratories in the area were generally prepared to do the work required for the Program. However, QA/QC activities during the Variability Phase, including review of raw data sheets and site inspections, revealed poor performance by one laboratory representing several dischargers. Regional Board staff responded by requiring many test repeats. The justifications for test repeats were: 1) missing data for entire effluent tests, 2) missing data for reference toxicant tests, 3) reference toxicant data that repeatedly showed no dose response, 4) inadequate performance in control treatments, and 5) inadequate efforts to obtain spawning animals for urchin fertilization and mollusc embryo tests.

The technical questionnaire soliciting comments on the test protocols received few responses, indicating a general confidence in the toxicity test methods. However, there were some comments that EPA protocol specifications could be interpreted in more than one way, or that protocols were not specific enough on subjects pertaining to the complex salinity adjustments required to conduct these studies in San Francisco Bay. Recommendations and requirements resulting from this questionnaire were distributed to program participants in a September 1,1989 mailing.

Results of the Screening Phase are fully described in Anderson (1989). Acute toxicity was observed in more than half of the effluents (five municipalities and six industries) tested during the screening phase. Three of the municipal and five of the industrial effluents were acutely toxic to more than one species. For both municipalities and industries, the average number of species affected was two.

However, the acute tests were generally not as sensitive as the chronic tests, as judged by the NOEC values. For industrial dischargers, acute tests were never among the three most sensitive tests. For municipal dischargers, the sanddab acute test was the most sensitive test for four sewage treatment plants with secondary treatment. Each of these plants conducted Toxicity Reduction Evaluations (TRE's), which determined that the sanddab toxicity was due, in large part, to high levels of unionized ammonia. The other acute tests were generally not among the three most sensitive tests conducted during the screening phase.

The order of sensitivity of the species used in acute testing was determined by ranking the percentage of significant (LC50 ≤ highest effluent concentration tested) responses observed for each species (Table 1). During the Screening Phase, diatom tests were sometimes considered acute tests and sometimes considered chronic tests. The sensitivity ranking for species tested more than ten times was: sanddab = diatoms > mysid shrimp > rainbow trout > water flea.

Sensitivity of acute tests was next evaluated by ranking the number of sites for which a species exhibited the lowest acute effect level (LC50). This method gave the same ranking as the previous method (Table 1). All of the effluents for which sanddab and *Neomysis* were observed to be the most sensitive species were municipal effluents.

Chronic toxicity was detected in all but three of the effluent tests. Fourteen of the fifteen effluents which exhibited chronic toxicity were toxic to more than one species. For both the municipalities and the industries, the average number of species affected was three.

Species sensitivity in chronic testing was first ranked according to the percentage of significant (NOEC < highest effluent concentration tested) responses for each species (Table 2). Only tests conducted more than five times were included in this analysis. The ranking was: water flea > sea urchin > mussel embryos > fathead minnow = diatoms > silverside minnow. Sensitivity was next ranked according to the percentage of sites for which a species gave the lowest effect level (lowest NOEC) for a given site (Table 2). The ranking was: silversides > sea urchin = mussel > water flea > fathead minnow > diatom. This evaluation indicated that the lowest ranked species in terms of percentage of significant responses (the silverside minnow) was usually the most sensitive when it showed a significant response. Conversely, the species that responded most frequently (water flea) was not among the most sensitive when it showed a significant response.

Results of the Variability Phase, based on six-month progress reports, are fully described in Anderson et al. (1990a). Based on Regional Board review of these reports, toxicity reduction evaluations (TRE's) may be beneficial at several sites, including two oil refineries and one sewage treatment facility with advanced treatment. A TRE is ongoing at one oil refinery, and one sewage treatment facility has completed a TRE. At most sites, however, additional information is needed, either to better define initial dilution at the discharge site or to verify observed effects.

Ambient Toxicity Program

The results of the Ambient Toxicity Program surveys are fully described in Anderson et al. (1990a, 1990b). Briefly toxicity was documented in marshes and Bay "background" stations, located at a distance from point sources of pollutants. Chronic toxicity was observed at three locations within the National Wildlife Refuge, which receives only urban runoff. Toxicity was also observed at many of the twelve Bay "background" stations during all four surveys, using the echinoderm fertilization test. No toxicity was observed using silverside minnow or mollusc development tests.

At Hayward and Mountain View marshes, created from undiluted effluent, toxicity was observed at several locations. Toxicity was most pronounced at Hayward marsh, where acute toxicity was observed for two of the three species tested during two separate surveys, with the effects distributed over approximately half of the basins in the marsh closest to the discharge. A preliminary Toxicity Reduction Evaluation was conducted, and it was determined that at least a portion of the toxicity in the marsh was attributable to unionized ammonia.

A survey of the marsh receiving up to twenty-three million gallons per day of advanced treated wastewater from the city of Sunnyvale showed chronic toxicity in the vicinity of the discharge. The results of the ambient toxicity survey confirmed predictions from effluent tests concerning levels of toxicity (no toxicity vs. chronic vs. acute) and the species that would respond. Chronic toxicity was observed in both effluent and ambient samples using *Ceriodarhnia*, urchin fertilization and diatom tests; no toxicity was observed in effluent or ambient samples using fathead minnow. In contrast, acute toxicity was observed using *Ceriodarhnia* in a storm drain sump that discharges into the same area during wet weather. In addition, there was no observed toxicity using diatoms in an ambient sample taken from a downstream tributary to the discharge.

A survey of the marsh receiving 120 mgd of advanced treated sewage from the cities of San Jose/Santa Clara revealed no toxicity at any of the ten stations to a distance of five miles from the discharge. The survey near the USS Posco steel refinery discharge indicated that toxicity was associated with both intake and discharge waters. Toxicity could not be specifically attributed to the treated wastes produced by the refinery. Significantly acute toxicity, was detected in intake water from the Contra Costa Canal, a conveyance system that also supplies drinking water to the county.

TABLE 1. Percentage of significant responses observed for each acute test conducted during the Screening Phase, and the number of sites at which an acute test was the most sensitive (lowest LC50). Values in parentheses are the total number of times tested. The results for tests conducted > 10 times are listed separately from results of tests conducted ≤ 10 times.

	Percent Response	No. of Sites Most Sensitive
Species Tested > 10 times:		
Citharichthys stigmaeus (sanddab) (16)	38	3
Skeletonema costatum, Selenastrum carricornutum or *Thalassiosira pseudonana* (17)	38	3
Neomysis mercedas (mysid shrimp) (14)	36	2
Oncorhyncus mykiss (rainbow trout) (16)	25	1
Daphnia magna (water flea) (14)	14	1
Tested 2-10 times:		
Photobacterium phosphoreum (microtox bacterium) (4)	50*	0
Mysidopsis bahia (mysid shrimp) (2)	50	0
Neanthes arenaceodentata (polychaete worm) (6)	17	0
Palaemon macrodactylas (korean prawn) (10)	0	0
Menidia beryllina (silverside minnow) (2)	0	0

* The percentage of microtox responses may actually be lower, because results were not reported by some dischargers who conducted the test optionally.

TABLE 2. Percentage of significant chronic responses (number of significant responses divided by total number of tests conducted), and percentage of sites for which a species gave the lowest chronic effect level (NOEC). The values in parentheses are total number of times tested.

	% Response	% Most Sensitive
Menidia beryllina (silverside minnow) (16)	44	57
Strongylocentrotus purpuratus (sea urchin) (13)	77	50
Mytilus edulis (mussel) (11)	73	50
Ceriodaphnia dubia (water flea) (14)	79	27
Pimephales promelas (fathead minnow) (14)	50	14
Selenastrum capricornutum, *Skeletonema*, or *Thalassiosira pseudonana* (diatom) (13)	50	0

DISCUSSION

The Regional Board's effluent and ambient toxicity programs are providing data for assessing the risks of discharging toxic wastes into San Francisco Bay. The Effluent Characterization Program is currently generating No Observed Effect Concentrations (NOEC's) for twenty municipal and industrial discharges. Toxicity Reduction Evaluations (TRE's) are required when a significant risk to the estuary is indicated by NOEC's that are less than instream waste concentrations (IWC's) of the discharge. To date, data have indicated that toxicity reduction evaluations are warranted for one third or more of the twenty discharges tested in the program (Anderson et al., 1990a).

The Ambient Toxicity Program is providing data on toxicity associated with marshes and open water "background" stations (Anderson et al., 1990b). The surveys have been a useful tool for verifying risk assessments based on effluent toxicity testing and dilution assessments. In three cases, ambient surveys conducted in the vicinity of discharges verified predictions based on effluent testing by correctly predicting the level (no response, chronic or acute response) and the species that responded.

The ambient toxicity surveys documented acute toxicity in a marsh created from secondary treated sewage, a storm drain and a conveyance supplying drinking water. Chronic toxicity was documented at other marshes, including a natural marsh within the National Wildlife Refuge, Bay "background" stations, a slough supplying intake water for a steel refinery and a dredging site. No toxicity was observed in a marsh receiving 120 million gallons per day of municipal wastewater (secondary treated with nitrification and filtration).

These results support the view that wetlands used for wastewater treatment must be carefully evaluated and monitored for effects of toxic pollutants. It is still unknown if toxic pollutants in these systems pose more harm than benefit. Also, these observations underscore the importance to our Region of addressing both point and non-point sources of toxic pollutants in the Toxicity Control Program.

An interesting finding was that, with few exceptions, toxicity at Bay "background" stations was only observed using the echinoderm fertilization assay. During the first of four surveys, almost complete inhibition of fertilization was observed. One interpretation of this finding is that moderate toxicity persists throughout the Bay with more extreme toxicity occurring periodically. The widespread nature of the toxicity, based on the response of only one species, is perplexing but not unprecedented. In the Sacramento and San Joaquin Rivers, toxic effects have been documented along river stretches up to 75 miles in length using ambient toxicity testing (Foe and Connor, 1989). Only one (*Ceriodaphnia*) of several species tested showed a toxic response.

The second possible interpretation of these data is more speculative, but cannot be ruled out in a complex system such as the San Francisco Bay and Delta. There may be positive interferences in the application of this test to Bay waters, such as toxic effects of substances excreted by marine bacteria, physical effects of colloidal material, or subtle changes in ionic content of sample waters. Possibilities of this nature could be addressed experimentally.

The effluent and ambient toxicity programs have not only provided data to evaluate toxicity in discharges and Bay waters, but have also demonstrated that short-term, chronic toxicity tests, including relatively complex marine tests requiring salt additions, can be routinely performed by commercial laboratories. Inter-laboratory test results for chronic tests, obtained during an initial quality assurance round, demonstrated that test precision was comparable to the precision of chemical analyses, despite the fact that laboratories used different stocks of organisms and dilution waters (Anderson and Norberg, 1990).

However, the poor performance of one laboratory during the Variability Phase of the program indicates that QA/QC activities should continue beyond the initial step of determining the general preparedness of laboratories to conduct the tests required by the program. When contracting with commercial laboratories, dischargers should be aware of the additional challenges faced by laboratories conducting large volumes of toxicity tests.

Dischargers should review test results and supporting documentation, including raw data sheets, as soon as possible after the tests are completed.

Data from the Screening Phase of the Effluent Characterization Program have shown the importance of including a broad range of tests during an initial screening to determine the most sensitive test for each effluent. For the most part, the relative sensitivities of the acute and chronic tests could not have been predicted, based on a general characterization of the discharge, such as municipal vs. industrial or secondary vs. advanced secondary treatment with nitrification (Anderson et al., 1989).

Screening Phase data have also shown that discharges may be acutely toxic to one or more species, even though biomonitoring required by NPDES permits does not detect this toxicity. Our Region's required monitoring involves flow-through testing of a choice of two, out of three, fish species (fathead minnow, stickleback, and rainbow trout). These screening data will be used to re-evaluate the effectiveness of the present acute toxicity monitoring requirements in predicting the potential for acute toxicity in receiving waters.

Two general conclusions can be drawn from the Screening Phase data regarding the relative sensitivity of acute vs. chronic tests and sensitivities of the acute tests, when compared with one another. First, the acute tests were never among the three most sensitive tests (from a field of six acute and five chronic tests) for industrial discharges (all of the algal tests were categorized as chronic tests for this analysis). Second, acute tests were never among the three most sensitive tests for municipal dischargers, with the exception of the sanddab acute test, which was the most sensitive test to several secondary treated effluents. Based on these results, the Program may be revised to eliminate all acute tests from the Screening Phase. The sanddab test may, however, be required for some dischargers in compliance monitoring.

Toxicity reduction evaluations conducted in response to the observed acute toxicity to sanddabs indicated that the cause was, primarily, ammonia. However, these evaluations also demonstrated that methods used in conducting this test contributed to levels of unionized ammonia that would not be expected in receiving water. More specifically, the process of mixing hypersaline brine with effluent causing an upward shift of pH. This, in turn, increased the proportion of unionized ammonia in the test containers. One discharger also reported, however, that sanddabs responded to levels of unionized ammonia that were lower than those reported for other sensitive organisms cited in the EPA ammonia criterion document. The sanddab acute test may, therefore, be a sensitive tool for tracking ammonia toxicity problems, if they exist, in estuarine receiving waters.

Since few dischargers have submitted final reports, it is not yet possible to fully analyze Variability Phase data. The data will be used not only to evaluate the variability of effluents, but also to assess variability in test precision. Of particular interest is how precision of the echinoderm fertilization and mollusc larval abnormality test may be affected by reproductive seasonality. These analyses will be important in establishing reasonable specifications (e.g., monitoring frequency, species, dilutions, etc.) for permit limitations.

ACKNOWLEDGEMENTS

The San Francisco Bay Regional Water Quality Control Board is grateful for the excellent cooperation from municipal and industrial facilities participating in these Programs, the East Bay Regional Park District and the San Francisco Bay National Wildlife Refuge. Erika Hoffman, David Steward and Lisa Palermo and contributed substantially during many stages of this work. Numerous Regional Board staff assisted with field sampling. The Ambient Toxicity Program has been funded by the United States Environmental Protection Agency under an agreement with the California State Water Resources Control Board.

REFERENCES

Anderson, S. and T. J. Norberg. 1990. Editorial: Precision of short-term toxicity tests in the real world. Environ. Toxicol. Chem. in press.

Anderson, S. L., W. S. Pease, M. P. Carlin and T. E. Mumley. 1987. Proposed guidelines for the Effluent Toxicity Characterization Program, San Francisco Bay Regional Water Quality Control Board (8/87). Oakland, CA.

Anderson, S. L. 1989. Effluent Toxicity Characterization Program. Report to the San Francisco Bay Regional Water Quality Control Board (7/19/89). Oakland, CA.

Anderson, S. L., M. P. Carlin, L. Suer. 1990a. Status of the effluent and ambient toxicity characterization programs. Report to the San Francisco Bay Regional Water Quality Control Board (3/20/90). Oakland, CA.

Anderson, S. L., E. Hoffmann, D. Steward, and J. Harte. 1990b. Ambient toxicity characterization of San Francisco Bay and adjacent wetland ecosystems. Report to the California Water Resources Control Board (9/90). Sacramento, CA.

Foe, C. and V. Connor. 1989. Rice season toxicity monitoring results. Memorandum of the Central Valley Regional Water Quality Control Board (3/29/89). Sacramento, CA.

ENHANCEMENT TO THE PIELOU METHOD FOR ESTIMATING THE DIVERSITY OF AQUATIC COMMUNITIES

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ABSTRACT

The Pielou method of estimating the diversity of aquatic communities by plotting the relative abundances of the three most common species per sample on ternary plots, has been further developed. Algorithms are presented which permit the ternary plots to be readily produced via any computer spreadsheet having orthogonal graphics capability. Moreover it is shown that the Pielou method may be modified without loss of statistically significant information, so as to present time series diversity data as chronological tables or plots similar to those commonly used for other water quality variables, rather than as the ternary plots used by Pielou. Additionally the Pielou method has been extended from 3-dimensional to n-dimensional relationships. The usefulness of this new method for monitoring the impacts of toxicants to aquatic communities is demonstrated through examination of phytoplankton data for two oligotrophic lakes, one impacted by acid mine drainage and one pristine.

INTRODUCTION

Environmental scientists often describe and compare the diversity of biological assemblages or communities on a quantitative basis through use of various mathematical algorithms, and numerous papers have been published regarding species diversity indices and related algorithms including similarity indices, dissimilarity indices, richness indices, and evenness indices. Such indices are important to current ecological theory and are extensively used in environmental research; various papers offer comparisons between various indices and suggest criteria for selection (for example Lamont and Grant 1979; Wu 1982; Washington 1984; Sai and Mishra 1986;).

With regard to the aquatic environment, Pielou (1981) suggested that the great effort required to identify and count individual microscopic or taxonomically difficult organisms is prohibitively high with regard to the usefulness of the estimates achieved. Pielou recommends that greater effort be placed on collecting many samples from any water body being studied, with reduced effort being placed on the examination of each individual sample, and towards this end Pielou (1979, 1981) has developed a rapid method for estimating the diversity of each sample based upon the relative abundances of the three most abundant species within each sample. Pielou represented the species diversity of each sample as a single point on a ternary graph, and of each collection of samples as a swarm of such points. Visual inspection of the pattern taken by the swarm of "diversity points" on such a diversity plot yields a qualitative estimate of the diversity; all the points will fall within a right-angled triangle which Pielou called a "diversity triangle".

Where quantitative measure is required, Pielou recommended that the number of diversity points within each half area of the diversity triangle be counted and the ratio determined; where two communities need be compared, Pielou recommended that the number of points within each half area be counted, a 2 by 2 table set up and a x^2 test performed. Two benefits of the Pielou method of estimating species diversity are that comparisons can be made between communities having no common species, and that historical records where only most abundant species were identified and counted can now be used on a quantitative basis.

Pielou solely exemplified her technique with communities of marine benthic foraminifera. This present paper shows that this technique is also useful with regard to freshwater phytoplankton communities. Additionally, this paper presents algorithms so that the necessary ternary graphs may conveniently be drawn as routine orthogonal X-Y graphs, and describes how the abundance ratio method can be extended to permit diversity results to be presented both chronologically and for ratios other than ternary.

METHODS

Statistical Calculations

Following the Pielou method, for every sample the three most abundant species were determined, each set sequenced into descending order, and the count for each of these three divided by the summed count for the three, yielding q_1, q_2 and q_3 such that $q_1 + q_2 + q_3 = 1$ and $q_1 \geq q_2 \geq q_3$. The author has determined that since the "diversity triangle" defined by Pielou is a 30-60 right triangle, then any single point defined by the three abundance ratios ql, q2 and q3 can in fact be represented precisely by a single pair of X-Y coordinates, where $X = (q_2 + q_3/2)/0.86603$ and $Y = q_3$. Therefore ternary graphs were readily produced via standard orthogonal graphics computer packages, for example via an EXCEL spreadsheet such as illustrated in Figure 1. The vertices of the diversity triangle are (0,0), (0.5773,0) and (0.5773, 0.3333).

Pielou suggested that the diversity of two species assemblages can be compared statistically (i.e. null hypothesis that the two diversities do not differ) by dividing each diversity triangle into two halves of equal area, counting the number of points within each area, preparing a 2 by 2 matrix and performing a routine chi-squared test. Examination of the trigonometry shows that the half-area dividing line intercepts the diversity triangle at (0.1691, 0.0976) and (0.5773, 0.0976), and moreover shows that the statistical test proposed by Pielou is dependent solely upon Y-values and is entirely independent of X-values. Since the X-values are little meaningful, the author recommends that abundance ratio based estimates of diversity be presented not as ternary plots but rather as chronological tables or chronological graphs. This permits chronological patterns to be visually obvious and presents the diversity data in a format similar to that commonly used for other water quality variables. For statistical tests identical to that described by Pielou, points may be counted above and below the geometric half-divider $Y = 0.0976$, a 2 by 2 table set up and a x^2 test performed as adjusted for continuity.

Pielou restricted her consideration of abundance ratios to the three most abundant ratios. However, this concept may be extended to the n most abundant ratios: since the n^{th} abundance ratio value always lies upon a line perpendicular to the base, then it will always be true that $Y = q_n$. The half-divider points, as determined via Monte Carlo calculations, (single precision, 10000 iterations) yielded the following mid-array cut-offs for $n = 2$ to 5 respectively: 0.332, 0.153, 0.088 and 0.057; though these differ from geometric half-dividers (0.250 for $n = 2$; 0.0976 for $n = 3$), the situation is analogous to mean versus median, and results from the fact that all points within the diversity figures are not of equal probability. Statistical comparisons of identical data sets using each n and different half-dividers were performed to ascertain the relative impact each choice had on the power of both x^2 and Student's t tests to distinguish between different data sets.

Phytoplankton Data

Phytoplankton data from Buttle Lake (site 0130082) and from Lizard Lake (site E206283) were examined. Both are oligotrophic lakes on Vancouver Island and both are surrounded by second growth timber. Buttle Lake is the largest in a chain of lakes in the Campbell River Basin, has its level controlled by the Strathcona Dam on the downstream Upper Campbell Lake, and is fed by several major creeks. Lizard Lake is located approximately ten miles from Port Renfrew, sits on a bench between Harris and Lens creeks about two miles from the San Juan River, and has no major inlets. Since 1966 Buttle Lake has received metals from a nearby mining operation, though this loading has been greatly reduced in recent years due to remedial measures; conversely Lizard Lake has little anthropogenic influence other than a small campsite.

Phytoplankton are sampled in Buttle Lake as a biological monitor of the potential impact of contaminants from the mine, and in Lizard Lake as part of a long term program watching for any acidification of lakes. Details as to sampling, preservation and analysis of the phytoplankton samples and also as to the water chemistry of Buttle Lake are described elsewhere (Deniseger et al. 1986, 1990).

RESULTS AND DISCUSSION

Ternary Graphs

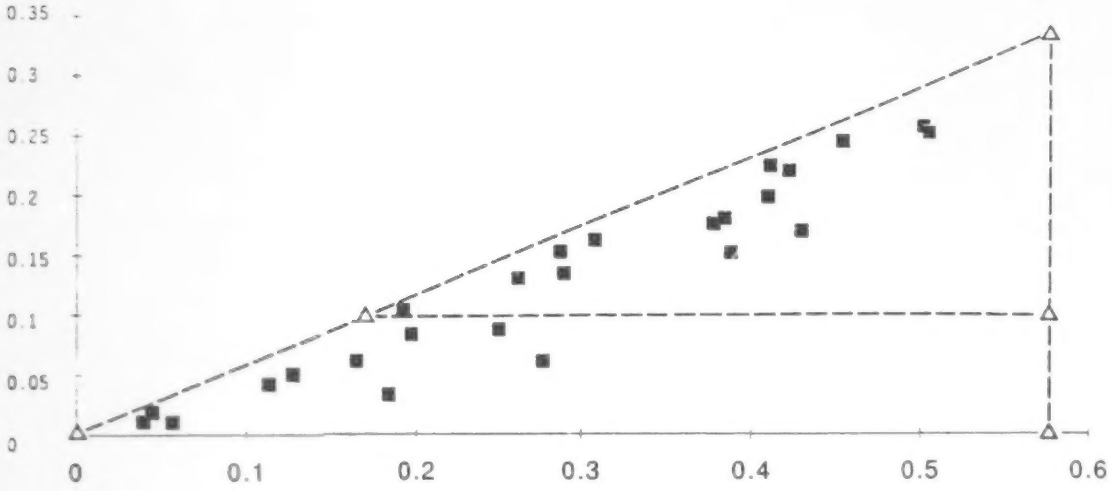
Ternary graphs as described by Pielou were prepared for the Lizard Lake and Buttle Lake phytoplankton data (Figure 2). The two plots appear similar except Buttle Lake has more points in the left vertex, i.e. one species tends to dominate the abundance. However when third most abundant ratio values are plotted chronologically, as in Figure 3, it is very apparent that species diversity has behaved very differently in the two lakes in that for Buttle Lake loss of species diversity for extended time periods occurred spring and early summer 1983 and again from summer 1984 to spring of 1986. These time periods correspond to the persistent bloom of the diatom *Rhizosolenia eriensis* (Smith) described by Deniseger et al. (1986, 1990). Lizard Lake showed loss of diversity for short periods of time, not any long intervals.

Similar patterns in species diversity for the two lakes are apparent when the n-most abundant ratio values are plotted chronologically, for $n = 2$ to 5, as presented in Figure 4. It may also be observed that as n increases, the graphs flatten, obscuring differences between periods of high and of low diversity for the graph scale used. To the eye, it does appear, as Pielou had suggested, that one need not go beyond the three most abundant species and indeed one might well argue that $n = 2$ is quite adequate.

BUTTLE LAKE PHYTOPLANKTON

DATE JULIAN YEARS	DATA				RATIOS				
	DOM. 1	DOM. 2	DOM. 3	SUM 1-3	"q1 = Dom1/Sum	"q2 = Dom2/Sum	"q3 = Dom3/Sum	X " = (q3/2+ q2)/0.866	Y " = q3
1966.62	14.70	4.50	3.80	23	0.639	0.196	0.165	0.321	0.165
1966.79	4.20	3.60	3.20	11	0.382	0.327	0.291	0.546	0.291
1967.62	28.80	15.90	9.60	54.3	0.530	0.293	0.177	0.440	0.177
1967.71	5.60	3.00	1.80	10.4	0.538	0.288	0.173	0.433	0.173
1980.63	26.20	12.20	6.40	44.8	0.585	0.272	0.143	0.397	0.143
1980.72	22.20	10.60	4.80	37.6	0.590	0.282	0.128	0.399	0.128
1980.80	25.80	7.00	6.60	39.4	0.655	0.178	0.168	0.302	0.168
1983.08	25.50	3.20	3.60	32.3	0.789	0.099	0.111	0.179	0.111
1983.23	110.40	6.80	2.40	119.6	0.923	0.057	0.020	0.077	0.020
1983.33	118.60	12.80	3.10	134.5	0.882	0.095	0.023	0.123	0.023
1983.42	731.00	85.70	6.20	822.9	0.888	0.104	0.008	0.125	0.008
1983.56	58.10	3.60	2.90	64.6	0.899	0.056	0.045	0.090	0.045
1983.65	19.00	13.50	10.90	43.4	0.438	0.311	0.251	0.504	0.251
1983.83	9.10	7.50	7.00	23.6	0.386	0.318	0.297	0.538	0.297
1984.13	53.80	31.70	5.30	90.8	0.593	0.349	0.058	0.437	0.058
1984.35	1157.00	206.50	38.50	1402	0.825	0.147	0.027	0.186	0.027
1984.48	2384.50	326.50	48.00	2759	0.864	0.118	0.017	0.147	0.017
1984.58	7334.50	326.50	19.00	7680	0.955	0.043	0.002	0.051	0.002
1984.65	2957.00	67.00	28.50	3052.5	0.969	0.022	0.009	0.031	0.009
1984.77	5385.50	31.00	10.00	5426.5	0.992	0.006	0.002	0.008	0.002
1984.96	9014.00	19.00	10.00	9043	0.997	0.002	0.001	0.003	0.001
1985.15	5061.00	28.70	19.10	5108.8	0.991	0.006	0.004	0.009	0.004

FIGURE 1. A portion of the Buttle Lake phytoplankton data shown with numbers rounded for presentation. The three columns to the right of the Julian-like date are the relative counts (no./mL) for the three most abundant species in each sample; the column titles indicate the calculations performed. The rightmost two columns, X and Y can be used to draw points in their relative locations for a ternary plot; alternatively the date and Y columns may be used to create chronological plots.



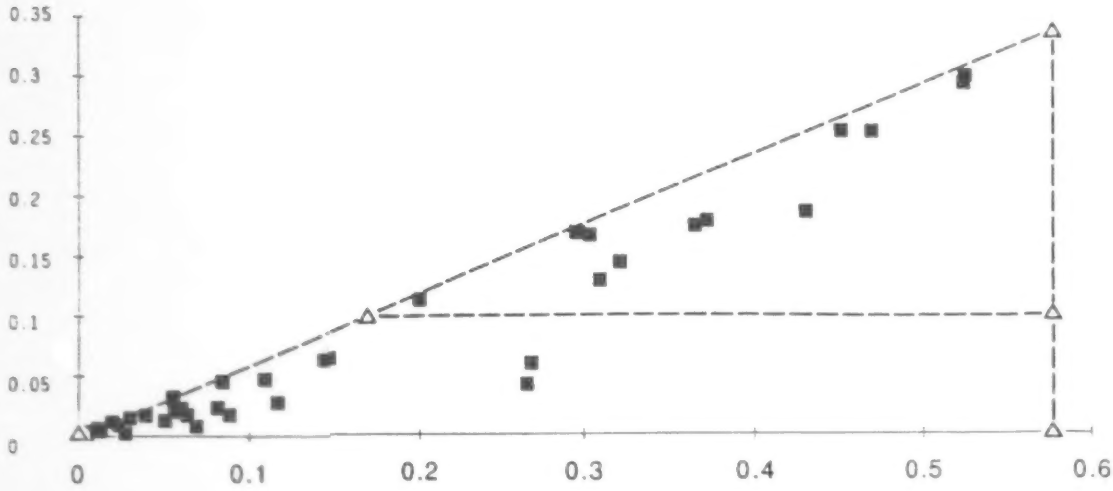


FIGURE 2. Pielou "diversity triangles" for Lizard Lake (top) and Buttle Lake (bottom) phytoplankton data. The data points are located in the relative positions they would have had on a ternary plot. The vertices of the diversity triangle are (0,0)), (0.5773,0) and (0.5773, 0.3333); the half-area dividing line intercepts the diversity triangle at (0.1691, 0.0976) and (0.5773, 0.0976)

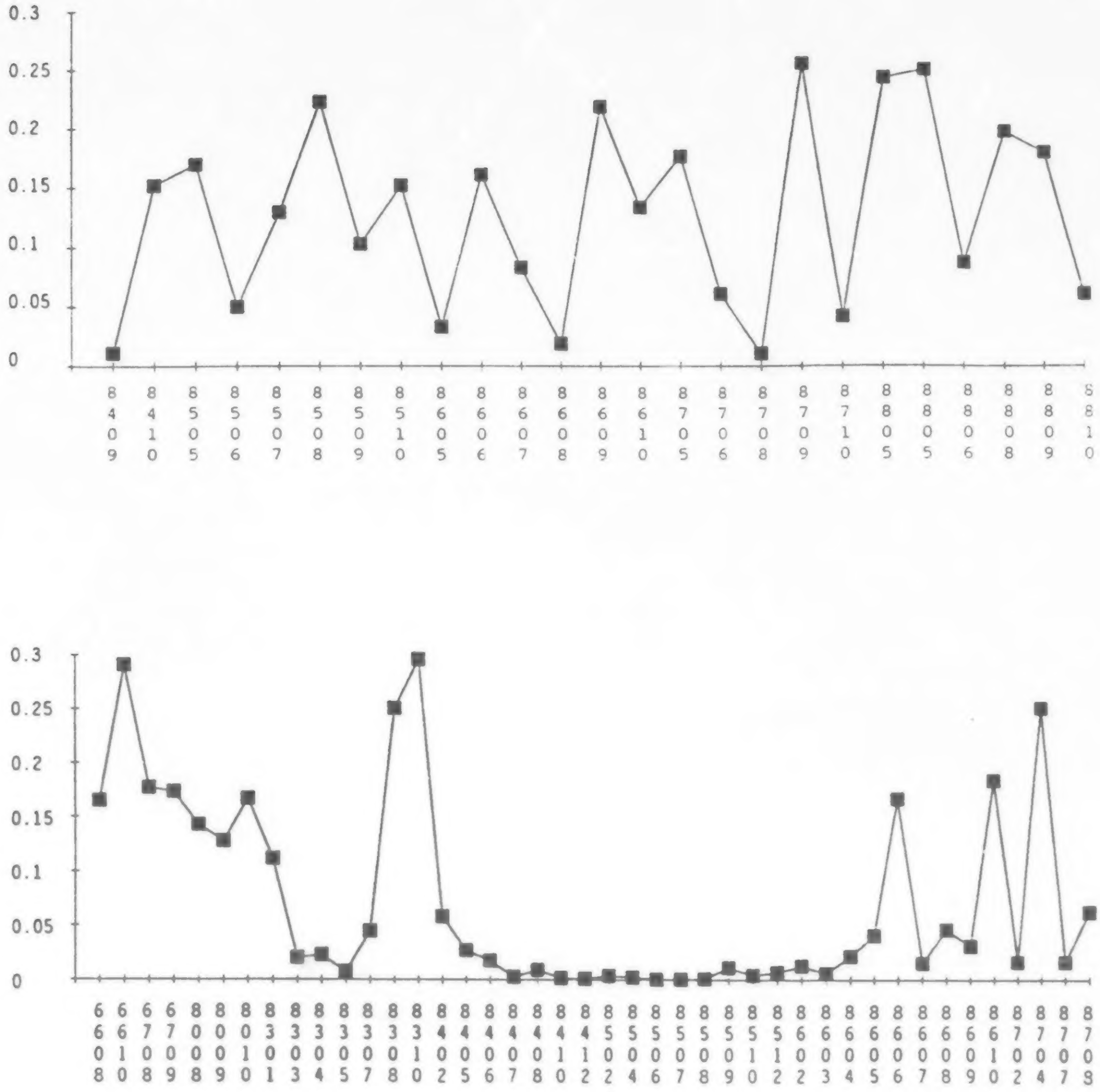


FIGURE 3. Chronological plot of species diversity for Lizard Lake (top) and Buttle Lake (bottom) phytoplankton data, as estimated from the abundance ratios of the three most abundant species, in each sample. For abscissa dates, 8409 is the 9th month of 1984. The Y-value for each point is identical to that it had when drawn on the Diversity Triangle, Figure 2.

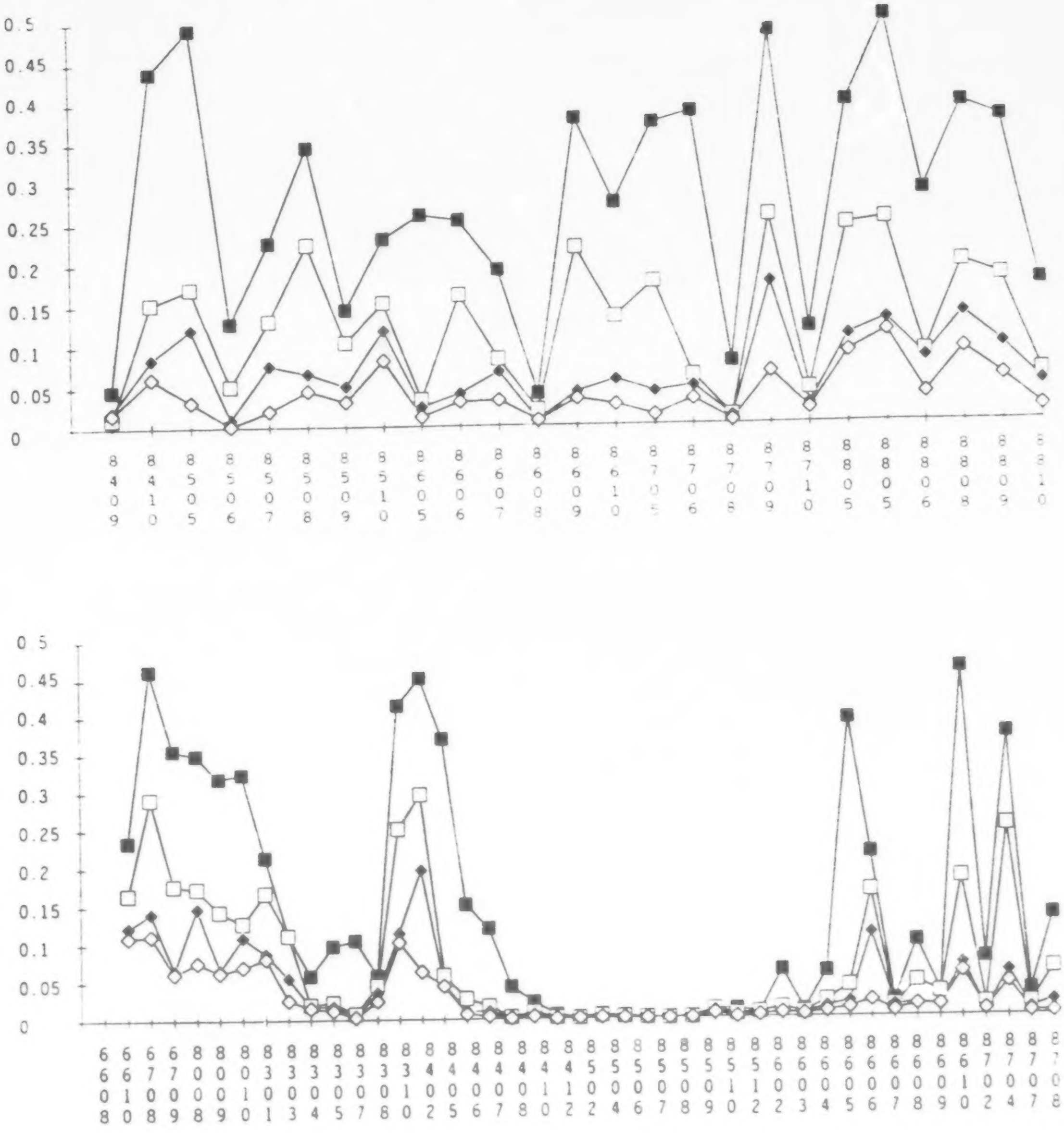


FIGURE 4. Chronological plot of species diversity for Lizard Lake (top) and Buttle Lake (bottom) phytoplankton data, as estimated from the two most abundant (open diamond), three most abundant (closed diamond), four most abundant (open square), and five most abundant (closed square) species in each sample. The Y-value for each point is the distance in (n-1)-dimensional space of the distance of the data point from the base of the "diversity figure".

X^2 tests, corrected for continuity, were unable at 95% confidence to distinguish any difference between the proportion of low to high diversity in phytoplankton data for the two lakes as estimated by the most abundant ratio method for n = 4 and n = 5, nor for n = 2 and n = 3 as estimated with the Monte Carlo half-dividers. However the x^2 test did show a significant difference at 95% confidence but not 99% for both n = 2 and n = 3 as estimated with the geometric half -dividers.

Since the chronological plots for the Buttle Lake results show three periods of different diversity, namely higher diversity followed by lower diversity followed by higher diversity, with each of the periods about one third of the number of samples collected, it seemed worthwhile statistically to compare each of these sub-records to Lizard Lake diversity. For the first of the three periods of record the x^2 test showed no difference between the two lakes at 95% confidence for n = 2,3,4,5 and for both types of half--divider; however the x^2 test could prudently be applied neither to either the second or third periods of data nor to the sum of the two since the minimum counts in the least proportions were not ≥ 5. Two-tailed F tests showed that there was no difference in variance at 95% confidence between Lizard Lake results and Buttle Lake results either in part or entire. Two-tailed Student's t tests failed to show any difference between mean diversity for Lizard Lake and for the first third of Buttle Lake data for n = 2,3,4,5 and also failed to show any significant difference between the Lizard Lake diversity and the entire set of Buttle Lake results for n = 5. However all other Student's t test showed Buttle Lake to be significantly less diverse than Lizard Lake at 95% and often 99% confidence. Thus it would appear that even after Buttle Lake had recovered from the extended period of low diversity, that its mean diversity was still reduced.

In summary, the Pielou method can be conveniently utilized via computer spreadsheet, can be modified so as to yield chronological graphs, and does appear to yield useful insights as to species diversity of freshwater lake phytoplankton. For this rapid method of estimating diversity there seems to be little benefit in examining more than the most abundant two species per sample, n = 2; also the Student's t test for statistical comparisons seems more in keeping than x^2 test with the concept of a rapid test, and has the additional benefits both of avoiding the moot question of what type of half-divider to use, and of being able to handle data sets with very low counts in one of the proportions.The problem of whether the geometric or Monte Carlo half-divider should be used towards establishing proportions for the test was not convincingly resolved but it is noteworthy that the x^2 test and Student's t test gave consistent results where the geometric half-divider was used.

ACKNOWLEDGEMENTS

The author is grateful to Lloyd Erickson and John Deniseger of the Waste Management Branch, British Columbia Ministry of Environment, Nanaimo for making their phytoplankton data available and for providing a description of Lizard Lake, and to Dr. Alan Austin of the Dept. of Biology, University of Victoria for study area in his research laboratory.

BIOLOGICAL EFFECTS OF CONTAMINANTS IN HALIFAX HARBOUR SEDIMENT

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ABSTRACT

Halifax Harbour sediments contain heavy metals and organic contaminants. To assess the biological effects of these compounds in the sediment, seven sites spanning a range of known sediment types, water depths, metal and organic contaminant levels were chosen for a series of bioassay tests. Sediment toxicity was measured by the reduction of microbial luminescence in three Microtox tests (pore water, solvent extract and solid phases), by the percent survival in two amphipod species (*Rhepoxynius abronius* and *Corophium volutator*) and by the survival and change in biomass of juvenile polychaete (*Neanthes sp.*). Uptake of contaminants from sediment was assessed by the bioaccumulation test using a bivalve mollusc (*Macoma balthica*). The chronic effects of the sediment on vertebrates were assessed by histopathological studies on winter flounder (*Pseudopleuronectes americanus*) collected from the harbour.

The results of this study were used to evaluate the extent of contamination in Halifax Harbour sediment and to assess the potential value of a battery of laboratory sediment toxicity tests for marine sediment environmental assessment in the ocean disposal program administered by Environment Canada.

INTRODUCTION

For more than a century, Halifax Harbour has been subject to waste discharges from the cities of Halifax and Dartmouth. Untreated sewage discharge, stormwater runoff, river discharge, household and industrial discharges and atmospheric fallout have introduced contaminants into the water column and harbour sediment.

In 1987, a harbour clean-up agreement was signed between Federal Government of Canada and Provincial Government of Nova Scotia for the construction of a primary treatment plant to clean up Halifax harbour. In response to this clean-up project, numerous scientific studies initiated by Environment Canada, Fisheries and oceans Canada and the Geological Survey of Canada were conducted in the harbour to assess the marine environmental quality of the harbour ecosystem. The results of some of these studies were published in a Canadian Technical Report of Fisheries and Aquatic Sciences (Nicholls, 1989). The report concluded that current state of knowledge of the marine environmental quality of the harbour is limited and that additional studies would be required to enhance our understanding of the harbour ecosystem. However, the harbour sediment chemical data provided by that report have confirmed the apparent polluted condition of the majority of the harbour area.

The present study was designed to assess the biological effects of the contaminants in harbour sediments. Based on surficial sediment chemical data reported by Buckley and Hargrave (in Nicholls, 1989) and Environment Canada (unpublished data), seven sites spanning a range of known sediment types, water depths, metal and organic contaminant levels were chosen for comparison by a series of bioassay tests. Sediment toxicity was measured by the reduction of microbial luminescence of a marine bacterium, *Photobacterium phosphoreum*, in three Microtox tests (pore water, solvent extract and solid phases), by percent survival of two amphipods collected from east and west coasts (*Rhepoxynius abronius*), and by the survival and change in biomass of juvenile polychaete (*Neanthes sp.*). The bioavailability of contaminants in the sediment were measured by uptake in a bivalve, *Macoma balthica*, and the chronic effects of the sediment on fin fish were assessed by histopathological studies on winter flounder, *Pseudopleuronectes americanus*, collected from the harbour.

The extent of contamination in the Halifax Harbour sediment will be assessed and the sensitivity and potential of each bioassay test conducted in this study will be evaluated to assess their effectiveness for the marine sediment environmental assessment in the ocean disposal program administrated by Environment Canada.

MATERIALS AND METHODS

Field Sampling

A 0.1 m^2 Van Veen Grab was used to collect sediments from the Bedford Oceanography Institute (BIO) research vessel "Sigma-T" at the seven sites (Table 1; Figure 1). Sediments were stored in clean 5 gallon plastic buckets,

TABLE 1. Coordinates for sampling stations in Halifax Harbour, N.S.

LOCATION	LATITUDE	LONGITUDE	DEPTH
1. Bedford Bay	44°43.1'	63°39.9'	12
2. Central Bedford Basin (#1)	44°41.15'	63°37.6'	44
3. Central Bedford Basin (#2)	44°41.10'	63°37.8'	60
4. Tuft's Cove (#1 and #2)	44°40.65'	63°35.95'	4
5. Imperoyal Jetty (Imperial Oil)	44°38.3'	63°32.8	12
9. Eastern Passage Sewage Treatment Plant	44°37.7'	63°31.5'	10
7. Drake's Gut	44°36.3'	63°30.35'	5

covered and transported to the Environment Canada Regional Laboratory located at BIO. Subsamples for particle size and chemical analyses were obtained after the sediments were thoroughly mixed in the buckets. The remaining sediment was stored in a refrigerator at 4°C until used in experiments.

Sampling was conducted at Bedford Bay, Central Bedford Basin, Tuft's Cove, Eastern Passage Treatment Plant Outflow and Drakes Gut on June 28, 1990. These sediments were used for the first set of bioassay tests (Test #1: Microtox Test, Amphipod Bioassay, Polychaete Bioassay and *Macoma* Bioaccumulation Test) which was carried out between July 4 and July 24, 1990.

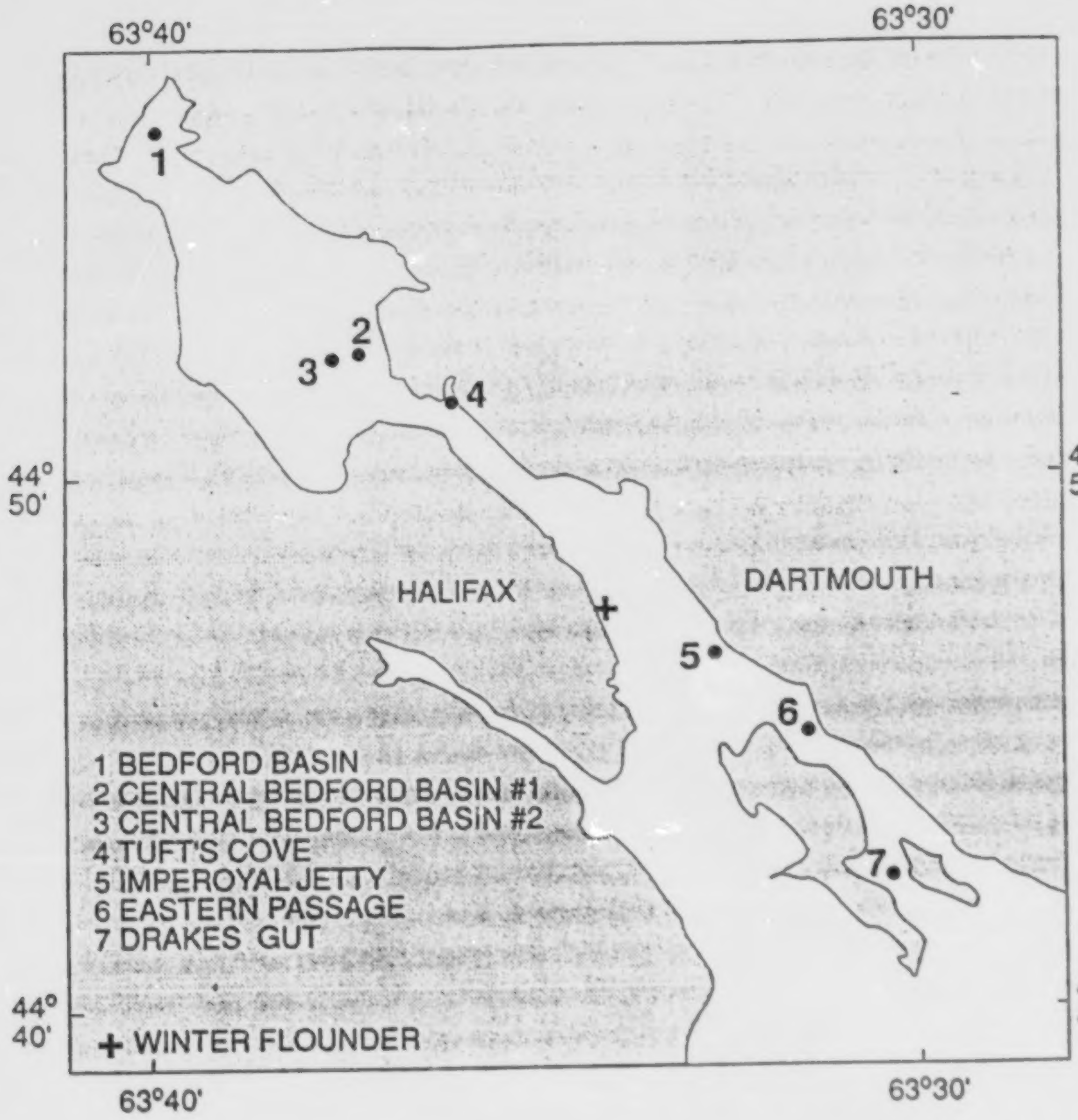
For the second set of bioassay tests (Test #2: Microtox Test, Amphipod and Polychaete Bioassays), sediments were collected from Central Bedford Basin, Tuft's Cove and Imperoyal on August 16, 1990. These tests were conducted between August 20 and September 12, 1990. In addition to the control sediments, a sediment of known toxicity collected from Sydney Tar Pond, Sydney Harbour, was used in Test #2 to assess the sensitivity of the test organisms.

Sediment Particle Size and Chemical Analyses

Particle size, inorganic and organic carbon analyses were conducted under a contract by Fenwick Laboratories Limited. Particle size analysis was performed by sieve and pipette following the method recommended in the Environment Canada Ocean Dumping Report #1 (Walton, 1978). Total organic matter and inorganic carbon (CaCo$_3$) were determined by loss-on-ignition at 420°C and 850°C respectively (McKeague, 1978).

Sediment and tissue heavy metals analyses were performed by the Environment Canada Regional Laboratory. Different digestion methods were used for metal extraction (Cd, Pb, Zn, Cu by total destruction with HF; Hg by sulfuric acid and potassium persulfate; As by sulfuric and nitric acid). Cd concentrations were obtained using flameless heated graphite furnace while Hg concentrations were measured by flameless atomic absorption. Pb measurements were obtained by flame atomic absorption and Zn, Cu and As were measured by ICP atomic emission spectrophotometry (O. Vaidya, pers. comm.).

Polynuclear aromatic hydrocarbons (PAH) were analyzed by both the Environmental Protection (EP) and Inland Water Laboratories of Environment Canada. Analyses of PAH in sediment and tissue samples were conducted according to an in-house EP standard method. Briefly, an undried sediment or tissue sample was subjected to ethanolic – KOH saponification. The diluted digest solution was solvent partitioned and the concentrated extract was eluted from an adsorptive alumina column in order to obtain a PAH fraction. This was further purified by gel permeation chromatography prior to HPLC analysis using external standard techniques. Analytical quality control was ensured by analyzing reagent blanks with each batch of processed samples, by spiked column recoveries, and by replicate analysis of standard reference material (P. Hennigar, pers. comm.).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HALIFAX
DARTMOUTH
1 BEDFORD BASIN
2 CENTRAL BEDFORD BASIN #1
3 CENTRAL BEDFORD BASIN #2
4 TUFT'S COVE
5 IMPEROYAL JETTY
6 EASTERN PASSAGE
7 DRAKES GUT
+ WINTER FLOUNDER

FIGURE 1. Location of sediment and winter flounder sampling stations.

Sediment Toxicity Bioassays

Three sediment bioassays (Microtox Bioassay, Amphipod Bioassay and Polychaete Growth Test) were selected for this study. The selection criteria were: (1) methods that employed test organisms from different trophic levels (2) test organisms that were sensitive to toxicity of contaminants, easily accessed and maintained in our laboratory; (3) methods employed different end-points for the assessment of biological responses; (4) availabilities of resources and equipment in our laboratory.

For the Microtox Bioassay, we chose the pore water and solvent extract methods which have been used extensively for sediment quality assessment (Schiewe et al., 1985; Williams et al., 1986; Ribo and Kaiser, 1987; Giesy et al., 1988; Ribo and Rogers, 1990; True and Heyward, 1990). We also selected a sediment solid phase method which is being developed by the Microbics Corporation to measure the direct impacts of sediment contaminants on luminescent bacteria (unpublished).

For the Amphipod Bioassay, we chose the West Coast species, *R. abronius*, because a standard method is available for this species. This test has been used extensively on the west coast of the USA for sediment toxicity testing (Swartz et. al., 1982, 1985, 1986, 1989; Chapman et al., 1984; Williams et al., 1986; Long et al., 1990). An East Coast species, *C. volutator*, was used in the same test to compare the sensitivity of the two species.

A polycheate bioassay using survival and change in biomass of juvenile *Neanthes sp.* was selected because polychaetes are one of the major groups of organisms in the benthic infaunal community and the end point used in this test is an important physiological measurement. This test has been developed by the U.S. Army Corps of Engineers and the Puget Sound Dredged Disposal Analysis (PSDDA) study and later adopted by the Washington Department of Ecology to be used as part of the state's marine sediments management program (Johns et al., 1990).

MICROTOX TOXICITY TESTS

Pore water Microtox test

Pore water was obtained from sediment samples by low-speed centrifugation (about 1200 rpm for approximately 45 min). Pore water salinity ranged from 31-34 ppt, close to the Bedford Basin Seawater value of 32.5, with the exception of the Sydney Tar Pond pore water, which had a salinity of 4 ppt. All the pore water tests except Sydney Tar Pond were conducted using Bedford Basin Seawater as diluent (and control), as recommended in the Microbics Methods Manuals (Microbics, 1989a,b). The Sydney Tar Pond sample was ionically adjusted to 22 ppt using MOAS (Microtox Osmotic adjustment solution). One or more potassium dichromate reference toxic tests were conducted with each sample run as a check on technique and reagent sensitivity. In addition, two centrifuged Bedford Basin Seawater blanks were run as a check on the sample preparation method.

Organic Solvent Extract

The method used for solvent extraction was similar to that described by True and Heyward (1990), as modified from Schiewe et al. (1985). The percent total solids of each sample was measured first in order that a consistent quantity of sediment could be extracted. Briefly, to 10 g (dry weight equivalent) of sample in a 600 mℓ glass beaker, sodium sulfate (20 – 200 g) was added (with occasional stirring) until water was essentially removed. 100 mℓ of methylene chloride was then added, and the mixture placed in a sonicator (Branson, model B-S2H) for 15 minutes. The liquid was then decanted off and additional extractions continued until the methylene chloride became clear (two to six times). The combined extract was concentrated to approximately 6 mℓ by roto-evaporation in a 35°C water bath, then transferred to a glass centrifuge tube and further concentrated in a 35°C bath to 1 mℓ under nitrogen gas. One hundred microliters of this extract was added to 3 mℓ of ethanol and the solution concentrated to 1.5 mℓ (at 35°C under nitrogen) to ensure total evaporation of the methylene chloride. Ethanol was then added to restore the volume to 3 mℓ. This was the solution used in the Microtox Test.

Tests on sediment solvent extracts were conducted using a protocol (supplied by Microbics™) for samples dissolved in organic solvent. The sample was first diluted to 1% in normal microtox diluent, making a 1% sample (10,000 ppm) in 1% ethanol. Further test dilutions were made in ethanol diluent so that all test concentrations contained 1% ethanol. These were run against a 1% ethanol-diluent control which compensated for the slight direct toxicity caused by the ethanol carrier.

All solvent extract samples were first screened for toxicity at 10,000 ppm. The samples were then diluted or concentrated as required to bring them in range and then run according to the standard assay procedure (Organic Solvent Protocol) to determine 5 and 15 min. EC50 values.

One or more potassium dichromate tests were conducted with each sample run as a check on technique and reagent sensitivity. In addition, solvent extract procedure blanks were run to determine if any toxicity was being contributed by the added chemicals (e.g. sodium sulfate, methylene chloride) and glassware. As well, EC50's for double-distilled ethanol were run to determine how much ethanol toxicity the solvent extract procedure was in fact compensating for.

Solid Phase Assay

The solid phase assay was conducted by Microbics Corporation (unpublished method). Essentially, the bacteria are exposed to an aqueous suspension of the whole sediment, the sediment is removed by filtration, and light loss measured using a series of concentrations.

AMPHIPOD SEDIMENT BIOASSAYS

Corophium volutator

C. volutator were collected from the intertidal mud flats at Walton Beach, Bay of Fundy, Nova Scotia. To minimize injury, the animals collected were not sieved out of the mud. Surface mud was placed in water and shaken to release the animals from the sediment. A net was used to catch the dislodged animals.

Animals were transported in 5 – 10 litres of seawater back to the Environment Canada Regional Laboratory. Unsieved sediment was brought back to the laboratory to hold the amphipods. The animals were held in 1x1 metre tanks with flowing seawater and a layer of sediment on the bottom. They were slowly acclimated to 15 + 2°C and held at this temperature for a minimum of five days before the commencement of testing. Excess sediment was brought back to the laboratory and sieved through a 1 mm mesh to remove *C. volutator* and other macro-invertebrates. This sieved sediment was used as a control sediment during the toxicity tests.

Rhepoxynius abronius

R. abronius were supplied by EVS Consultants of Vancouver, British Columbia. They were collected from an uncontaminated, sub-tidal site by Whidbey Island, Washington State, United States. After collection, the amphipods were separated from sediment by sieving the sediment along with other organisms and debris and were then reintroduced to the sediment before they were brought to Vancouver in plastic containers. The capped containers were packed in a cooler with ice and flown overnight to Halifax arriving the next day. The amphipods were transported to the Environment Canada Regional Laboratory where they were gradually brought to 15 + 2°C in a continuous flow of seawater. They were held for a minimum of four days at this temperature before the commencement of testing.

TESTING PROCEDURES FOR AMPHIPODS

Testings for both species of amphipods were conducted following the Draft Environment Canada Aquatic Biological Test Method "Ten-Day Test for Sediment Toxicity using the Marine Infaunal Amphipod *Mepoxynius abronius*" (Environment Canada, 1989).

Five replicates were set up for each sediment tested. Each replicate contained 20 amphipods. The number surviving in the five control sediment replicates was compared with those recorded in the five replicates for each of the other sediments tested on the same date using an ANOVA test in the SAS™ microcomputer program. Significant differences at the 0.05 level between the control and treatment means were calculated by the Dunnett's T Test.

A 96-hour LC50 was performed without sediment for each amphipod species (ten animals per jar) using the reference toxicant cadmium chloride as a check for organism sensitivity.

Polychaete Sediment Bioassay

The Polychaete Sediment Bioassay was conducted following the method described by Johns et al. (1990). Juvenile polychaete, *Neanthes sp.*, were purchased from laboratory stock cultures of D. Reisch, California State University, Long Beach, CA. Five juvenile worms were added to 1 litre glass jars containing 2 cm sediment and filled with seawater (28 ± 2 ppt; 20 ± 1°C). Five replicate jars each containing five worms were set up for each sediment tested. Sieved sand from Whidbey Island, Washington State, was used as control sediment. Each jar was aerated through a glass Pasteur pipette. Eight mg of Tetramarin per animal was added as food every second day and one-third of the seawater was renewed every third day. Worms were exposed to the sediment for twenty days. At the end of the exposure period, worms were removed from the sediments and rinsed with distilled water. All surviving worms from each jar were placed on a preweighed drying pan and dried at 100°C for 24 hours. The animals were removed from the drying oven, allowed to cool in a desiccator, and reweighed. Survival, total biomass (dry weight) and mean individual biomass were the response criteria.

A reference toxicant test was performed without sediment using cadmium chloride. Data were analyzed as for the amphipod-tests.

Macoma Bioaccumulation Test

The Macoma Bioaccumulation Test was conducted according to a method developed by the Environment Canada Atlantic Regional Laboratory. The test was conducted only in the first set of bioassay tests (Test #1).

M. balthica (Baltic clam) were collected from Walton Beach, Nova Scotia. They were acclimated in sediment collected from Walton Beach and flowing Bedford Basin seawater for 9 days before the commencement of the testing. Sediment from Walton Beach (control), Drake's Gut (reference) and Tuft's Cove were used in the test. Sixteen litres of control water (Bedford Basin seawater) was added to each test tank followed by 4 litres of sieved (0.5 cm mesh screen) sediment. The sediment was then allowed to settle one day (with gentle aeration at 100ml/min) before placing *Macoma* (n = 60) on the surface of the sediment in each test tank. The test was conducted at 15 ± 1°C for an exposure period of 30 days. Each sediment was tested in duplicate and the tissues were pooled for chemical analysis. Temperature, dissolved oxygen, pH, and salinity were measured just prior to placing clams in the test tanks and every few days throughout the test period. Observations of clam burrowing and mortality were also made and recorded.

Tissue samples were taken from unexposed clams (n = 120) before the test, and exposed clams at the termination of the test. These clams were placed in running Bedford Basin seawater for 24 hours to allow for depuration of sediment before the tissue samples were removed. Tissue samples were rinsed with deionized distilled water, placed in whirl-packs and glass bottles and kept frozen for chemical analysis.

WINTER FLOUNDER HISTOPATHOLOGICAL STUDY

Three adult winter flounder (*Pseudopleuronectes americanus*) (Total length: 22.5 – 37.0 cm; Weight: 152.6 – 437.7g) were caught by fishing line from Queens Wharves in Halifax Harbour (Fig.1). Fish were held in a flow-through tank in the Halifax Fish Laboratory of the Department of Fisheries and Oceans for approximately two weeks before sacrifice. Length, sex and external abnormalities of each fish were recorded before tissue samples were taken from liver, kidney, gill and gonad for histological examination. Tissue samples (4 mm sections) were immediately fixed and preserved in 10% buffered formalin prepared by commercial source (Fisher). Samples were dehydrated, embedded in paraffin, sectioned at 5 microns, and stained with Harris' haematoxylin and eosin following routine protocols (Luna, 1968). Slides were examined systematically. Nine adult winter flounder collected from Georges Bank were used for control.

RESULTS

Particle Size and Chemical Analyses

Grain size distribution of sediments used in the toxicity tests ranged from the coarse substrate of Sydney Tar Pond sediments (68.6% gravel and 24% sand) to the very fine substrate of sediments from Bedford Bay (55.6% silt and 33.5% clay) and Central Bedford Basin (51.2% silt and 36.5% clay) (Table 2). The Whidby Island sediments where *R. abronius* were collected were mainly sand (98%) while all sediments from the Halifax Harbour with the exception of Drake's Gut were mostly silt and clay. The Tuft's Cove sediments used in Test #2 toxicity test had the highest percentage of clay (52.2%).

Total organic carbon of the tested sediments ranged from 0.77% in the Whidby sediment to 20.3% in the Sydney Tar Pond sample (Table 2). Organic carbon levels in the Halifax Harbour sediments ranged from 3.8% in the Drake's Gut sample to 8% in the second Tuft's Cove sediment sample. The control sediment from Walton Beach contained 3.98% organic carbon which was within the average levels (4.24 + 2.24%) found in Halifax Harbour sediments (Buckley and Hargrave, 1989).

Heavy metal concentrations were high in both the Sydney Tar Pond and Tuft's Cove sediments. High concentrations of mercury (1.6 – 22.9 ppm), cadmium (0.7 – 2.1 ppm) and

Station	As	Cd	Cu	Hg	Pb	Zn	PAH	TIC	TOC	GRAVEL	SAND	SILT	CLAY
				(ppm)						%			
Walton Beach	32	0.1	23	0.04	40	52	0.03	1.13	3.98	0	48.6	34.7	16.6
Whidbey Island	32	0.1	11	0.025	30	40	<0.01	0.24	0.77	0	98	0.3	2
Drake's Gut	15	0.6	25	0.08	46	71	0.77	4.6	3.8	0	51.7	39.8	8.5
Bedford Bay	50	0.4	47	0.32	89	150	0.51	4.6	4.7	0	10.9	55.6	33.5
Ctr. Bedford Basin #1	30	1.4	70	1	173	176	9.2	7.7	4.7	1.7	22.7	50.3	25.2
Ctr. Bedford Basin #2	64	0.1	121	1.6	288	370	8.49	9.13	4.53	1.1	11.3	51.2	36.5
Tuft's Cove #1	50	2.1	243	22.9	297	5580	3.32	19.1	6.4	2.7	21	47.1	29.2
Tuft's Cove #2	35	0.7	402	1.6	265	13584	25.43	25.1	8	4.6	19.2	24	52.2
Imperoyal	34	0.1	53	0.6	134	608	4.37	6.2	4.28	0	13	66.6	20.4
Eastern Passage	25	0.7	65	0.55	103	157	9.41	8.7	4	1	29.8	51.6	17.5
Sydney Tar Pond-100%	19	1.18	104	1.3	570	1625	6615.2	31.1	20.3	68.6	24	3.2	4.1
Sydney: Whidby 1:1-50%	38	0.2	43	0.4	205	595	2538.4	8.96	7.36	12.8	82.8	0.6	3.7
Sydney: Whidby 1:10-10%	37	0.3	16	0.08	58	126	434.8	1.95	2.38	2.5	93.9	0.5	3.1

TABLE 2. Particle size and chemistry data for sediments used in bioassay tests.

PAH	Walton Beach	Whidby Island	Drakes Gut	Bedford Bay	Ctr. Bedford Basin #1	Ctr. Bedford Basin #2	Tuft's Cove #1	Tuft's Cove #2	Imperoyal	Eastern Passage	50% Sydney Harbour	10% Sydney Harbour
1) LPAH												
Naphthalene	<0.01	<0.01	<0.04	<0.04	<0.01	0.22	<0.04	<0.02	<0.01	<0.2	367	65.4
Acenaphthene	<0.01	<0.01	<0.01	<0.1	<0.05	<0.006	0.07	<0.006	<0.006	<0.06	<0.047	<0.025
Acenaphthylene	<0.01	<0.01	<0.07	<0.7	<0.3	-	<0.09	-	-	<0.4	-	-
Fluorene	<0.01	<0.01	0.09	<0.2	0.19	<0.005	0.06	<0.005	0.19	0.46	85.4	14.6
Phenanthrene	0.02	<0.01	0.07	<0.1	1.1	<0.005	0.23	5.06	<0.004	0.99	484	91.8
Anthracene	<0.01	<0.01	0.03	<0.2	0.37	<0.02	0.08	6.99	<0.01	0.41	253	*IN
2) HPAH												
Fluoranthene	0.01	<0.01	0.11	0.21	1.2	2.32	0.46	4.49	1.13	1.3	386	74.6
Pyrene	<0.01	<0.01	0.09	0.13	1.1	2.53	0.43	1.96	1.45	1.1	330	66.6
Benzo(a)Anthracene	<0.01	<0.01	0.05	0.08	0.54	1.00	0.21	1.16	0.40	0.53	124	24.8
Chrysene	<0.01	<0.01	0.05	<0.2	0.64	<0.008	0.19	3.48	<0.007	0.52	144	34.6
Benzo(b)Fluoranthene	<0.01	<0.01	0.07	<0.09	0.66	0.96	0.36	0.97	0.51	1.1	100	19.3
Benzo(k)Fluoranthene	<0.01	<0.01	0.03	<0.04	0.33	0.18	0.13	<0.001	<0.001	0.6	53.1	10.1
Benzo(a)Pyrene	<0.01	<0.01	0.06	<0.09	0.73	1.28	0.28	1.32	0.69	0.68	116	21.7
Dibenzo(ah)Anthracene	<0.01	<0.01	<0.01	0.05	0.54	<0.002	0.12	<0.002	<0.001	0.41	12	<0.006
Benzo(ghi)Perylene	<0.01	<0.01	0.04	0.04	0.82	<0.005	0.23	<0.005	<0.004	0.46	26.5	<0.19
Indeno(1,2,3-cd)Pyrene	<0.01	<0.01	0.08	<0.2	0.98	<0.005	0.47	<0.005	<0.004	0.85	57.4	11.3
TOTALS:	0.03	<0.01	0.77	0.51	9.20	8.49	3.32	25.43	4.37	9.41	2,538.4	434.8

*IN - Interference

TABLE 3. Concentrations of PAH (ppm) of sediments used in Test #1 and #2.

TABLE 4. Summary of test results for three Microtox sediment assays

SAMPLE COLLECTION DATE	STATION	PORE WATER EC50 (%), 95%CL		SOLVENT EXTRACT EC5O (ppm), 95%CL		SOLID PHASE EC50 (ppm)
		5 min.	15 min.	5 min.	15 min.	
28 June	Solvent Blank	–	–	>20,000	>20,000	–
"	Walton (Control)	>100	>100	>20,000	>20,000	6728
"	Drake's Gut (Reference site)	>100	>100	474 (115-1948)	400	2460
"	Bedford Bay	>100	>100	867 (555-1352)	820 (574-1171)	1839
"	Central Bedford Basin	~90	~88	154 (83-286)	154	2384
"	Tuft's Cove	>100	>100	192 (72-512)	200	1986
"	Eastern Passage	~59	49 (22-109)	153 (73-320)	164	977
17 Aug.	Solvent Blank	–	-	>50,000	>50,000	–
"	Walton (Control)	>100	>100	–	–	–
"	Whidby Island (Control)	>100	>100	>50000	>50000	>28000
"	Imperial Oil	>100	>100	524 (448-613)	512 (213-1231)	1768
"	Central Bedford Basin	>100	>100	162 (119-219)	188 (103-344)	869
"	Tuft's Cove	11.6 (8.2-16.4)	11.8 (2.6-53.1)	58 (52-65)	69 (56-85)	1202
"	Sydney Tar Pond	7.4 (5.8-9.6)	8.8 (7.4-10.4)	13.5 (10.3-17.7)	16.8 (12.9-21.7)	1374
	10% Sydney Tar (in Whidby Island	12.7 (11.1-14.5)	15.2 (12.6-18.4)	261 (224-305)	263 (84-825)	2176

95%CL = 95% Confidence Limits

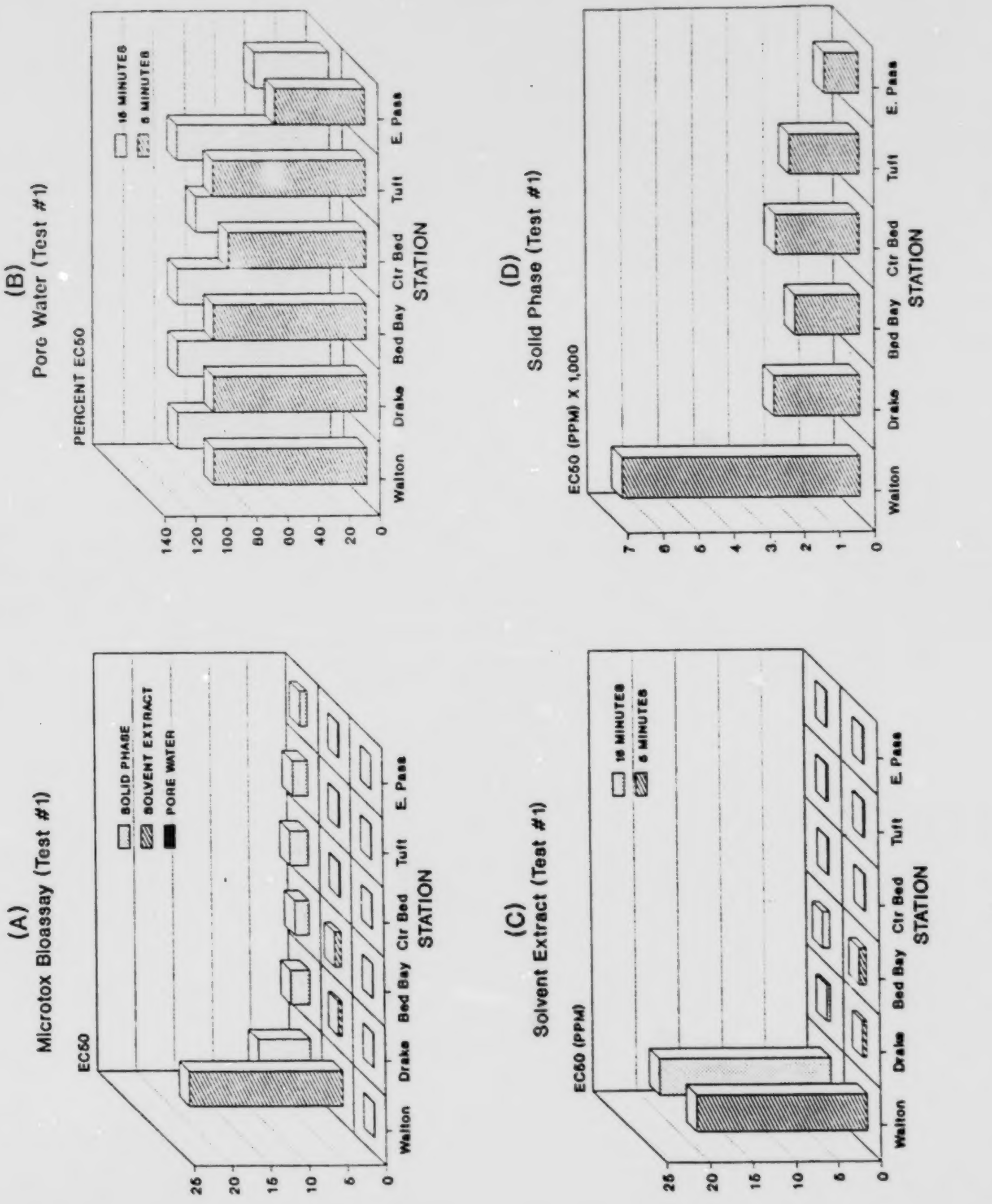


FIGURE 2. Results of Microtox bioassays in Test #1.

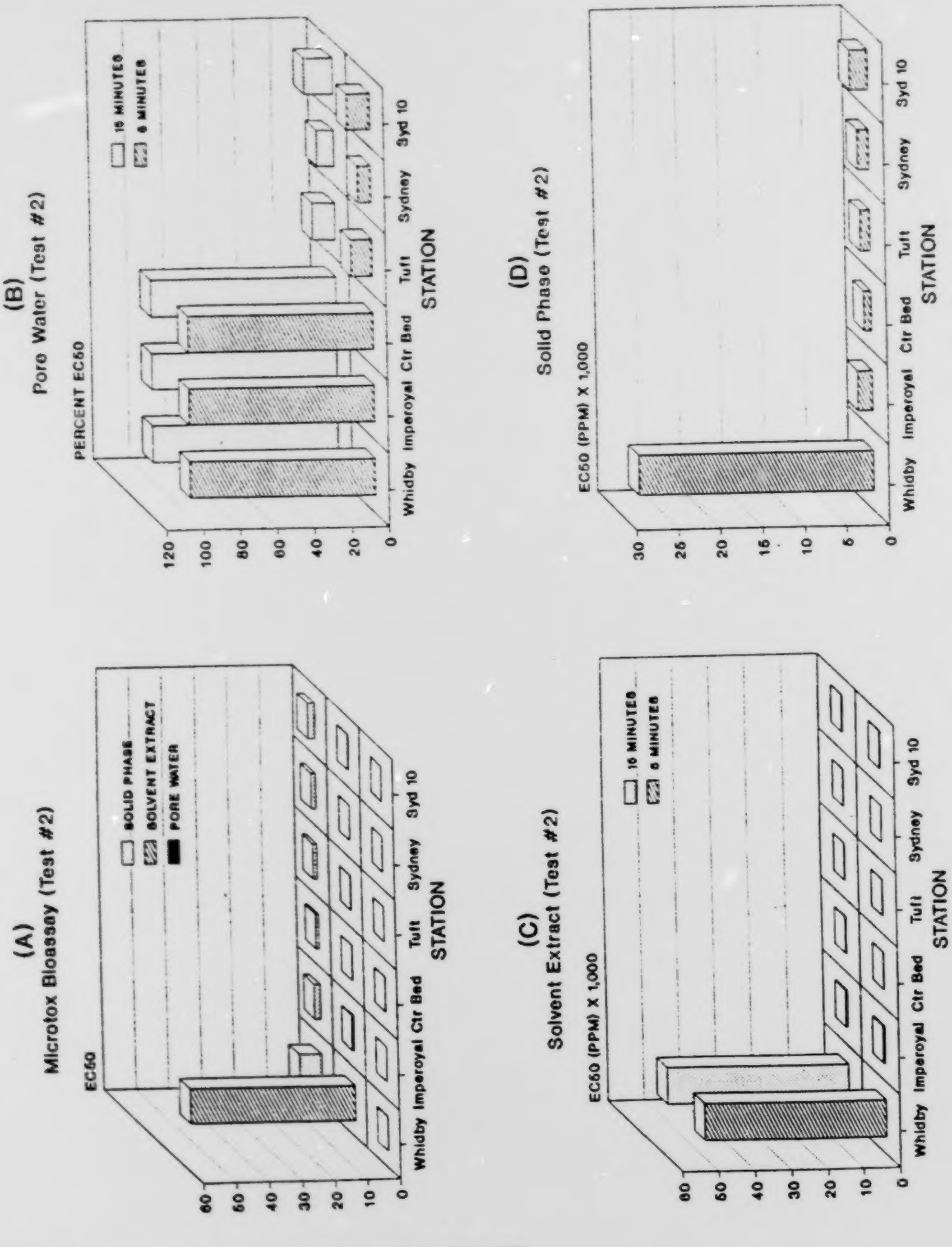


FIGURE 3. Results of Microtox bioassays in Test #2.

TABLE 5. Results of Amphipod sediment bioassays.

SAMPLE COLLECTION DATE	STATION	COROPHIUM VOLUTATOR		RHEPOXYNIUS ABRONIUS	
		% Survival (N=5)	% Burrowed at end of test	% Survival (N=5)	% Living animals that reburrow in clean sediment within 1 hour
28 June	Walton (control)	97 ± 4.5	Water cloudy	-	-
"	Whidby Island (control)	-	-	100 ± 0	100
"	Drake's Gut (Reference site)	97 ± 4.5	Water cloudy	95 ± 5	95.8
"	Bedford Bay	85 ± 20.6	Water cloudy	93 ± 5.7	92.3
"	Central Bedford Basin	97 ± 2.7	Water cloudy	79 ± 5.5 *	69.4
"	Tuft's Cove	87 ± 9.8	Water cloudy	96 ± 9.0	69.1
"	Eastern Passage	86 ± 8.2	Water cloudy	96 ± 4.1	82.5
17 Aug.	Walton (control)	96 ± 2.2	99	-	-
"	Whidby Island (control)	-	-	97 ± 2.7	100
"	Imperial Oil	99 ± 2.4	Water cloudy	44 ± 7.4 *	90.9
"	Central Bedford Basin	93 ± 5.0	97.3	47 ± 5.7 *	95.7
"	Tuft's Cove	91 ± 7.4	47.3	24 ± 21.0 *	62.5
	Sydney Tar Pond	0 ± 0 *	0 (All dead)	- **	- **
	50% Sydney Tar Pond (in control sediment)	0 ± 0 *	0 (All dead)	0 ± 0 *	0 (All dead)
	10% Sydney Tar Pond (in control sediment)	0 ± 0 *	0 (All dead)	0 ± 0 *	0 (All dead)

* Significant difference ($\alpha=0.05$) from control

** Interstitial salinity was 4 PPT. Rhepoxynius test not suitable at this salinity.

zinc (5580 – 13584 ppm) were detected in the two Tuft's Cove samples.The concentrations of polynuclear aromatic hydrocarbons (PAH) in Halifax sediments ranged from 0.51 ppm at Bedford Bay to 25.43 ppm at Tuft's Cove #2 station. The two control samples, Walton Beach and Whidby Island, contained 0.03 and <0.01 ppm of PAH respectively. The three Sydney samples contained very high levels of PAH (434.8 –6615.2 ppm) (Table 3).

MICROTOX TOXICITY ASSAYS

In the first set of bioassay tests (Test #1), of the 6 locations studied, inhibition of bioluminescence caused by pore water occurred only in sediment collected from Central Bedford Basin and Eastern Passage (Table 4, Figure 2). However, inhibition caused by solvent extract and solid phase of the sediment occurred in all sediments tested including samples from the reference site. In the second set of bioassay tests (Test #2), with the exception of the pore water test of Imperoyal and Central Bedford Basin sediments, the results showed that all the sediments were toxic to bacteria (Figure 3). Sediments from Sydney Tar Pond and the second Tuft's Cove sample had the greatest inhibitory effect on bacterial bioluminescence.

The control sediments (Walton Beach and Whidby Island) were consistently non-toxic.

Amphipod Sediment Bioassays

The percent survival of both amphipods in the control sediment for the two sets of bioassays ranged from 96% to 100%. *R. abronius* was generally more sensitive to the Halifax Harbour sediments than *C. volutator* (Table 5, Figure 4).

In Test #1, with the exception of the Central Bedford Basin sediment, none of the sediments from Halifax Harbour were toxic to either species of amphipod. The 79% survival of *R. abronius* in the Central Bedford Basin sediments was significantly lower than the % survival ($P < 0.05$) in the control batch. The number of *R. abronius* surviving that burrowed in clean sediment in less than a hour after the termination of tests was lowest in the Tuft's Cove and Central Bedford Basin sediments.

In Test #2, sediments from Sydney Tar Pond were acutely toxic to the two amphipod species. Sediments from the three Halifax sites were non-toxic to *C. volutator* but toxic to *abronius* ($P < 0.05$). The number of *C. volutator* burrowed in the test sediments at the end of tests (47.3%) and the number of *R. abronius* surviving that burrowed in clean sediment in less than a hour after the termination of tests (62.5%) were the lowest in the Tuft's Cove sediment.

Polychaete Sediment Bioassay

The Halifax Harbour sediments in Test #1 and Test #2 were non-toxic to the polychaete worms in the 20 days exposure experiment. No sublethal effects were detected in all exposures (Table 6; Figure 5). The 50% Sydney Tar Pond sediment was highly toxic to the polychaete larvae. The biomass dry weight measurements of worms surviving in 10% and 50% Sydney Tar Pond sediments (Test #2) were significantly ($P < 0.05$) lower than the biomass of worms surviving in the control and reference sediments.

Macoma Bioaccumulation Test

The percent survival of Macoma in the Tuft's Cove sediment was lower than that in the control (Walton Beach) and reference (Drakes Gut) sediments. No uptake of heavy metals and PAH were detected (Table 7, Figures 6 – 7).

Winter Flounder Histopathological study

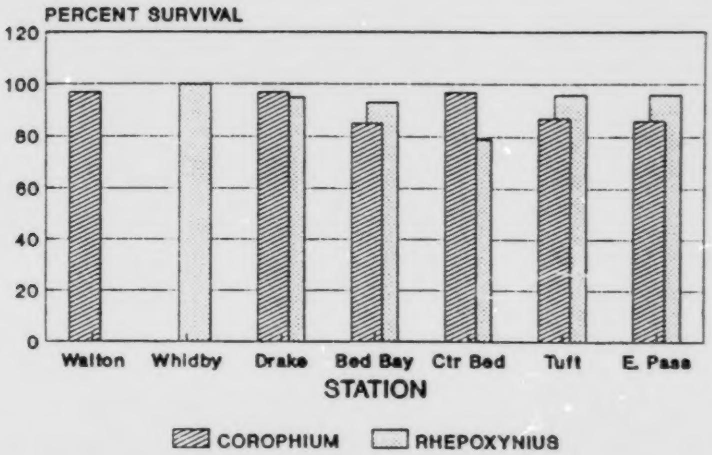
Histological examination of tissue samples of liver, kidney, testis and gills from the nine flounder taken from Georges Bank did not show any lesions.

Macrophage aggregation, hepatic epithelial vacuolation and hepatocyte basophilia were observed in the liver of winter flounder caught in Halifax Harbour (Table 8). The liver parenchyma was often intensely basophilic suggesting low lipid and glycogen stores. Mild early cellular vacuolation was detected, however, no biliary proliferation, vacuolation or neoplasia was seen.

A 1.5 mm focal lesion, with the characteristics of an immature seminoma was present in the testis of one of the Halifax flounder. Numerous spermatogonia were surrounded by nests of poorly differentiated cells.

The gill epithelium was hyperplastic in all Halifax flounder. Normal tubular and hemopoietic structure was seen in kidney of these fish, with many macrophage aggregates present.

(A)
Amphipod Bioassay (Test #1)



(B)
Amphipod Bioassay (Test #2)

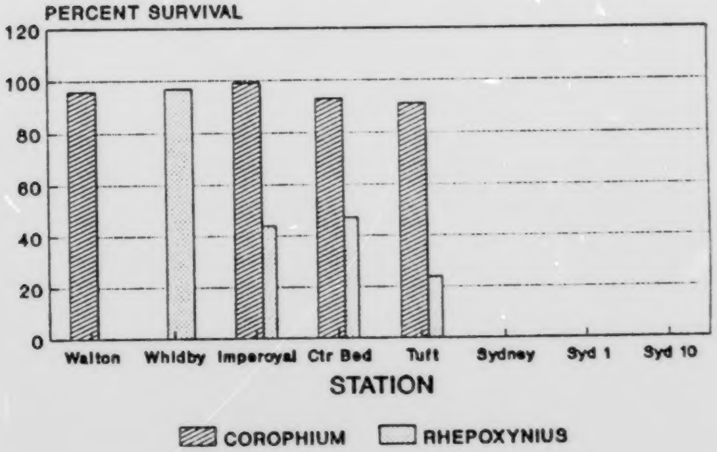


FIGURE 4. Results of Amphipod bioassys in Test #1 and #2.

TABLE 6. Results of *Neanthes* polychaete bioassays

SAMPLE COLLECTION DATE	STATION	% Survival (20 days) (N=5)	Average individual dry weight (g) per surviving worm (N=5)	Biomass per Chamber (g) (N=5)
28 June	Whidby Island (Control)	100 ± 0	0.03244 ± 0.00324	0.16216 ± 0.01619
"	Drake's Gut (Reference Site)	100 ± 0	0.03102 ± 0.00403	0.15508 ± 0.02011
"	Bedford Bay	100 ± 0	0.03454 + 0.00240	0.17266 ± 0.01198
"	Central Bedford Basin	100 ± 0	0.03032 ± 0.00248	0.15156 ± 0.01238
"	Tuft's Cove	100 ± 0	0.03343 ± 0.00336	0.16714 ± 0.01681
"	Eastern Passage	96 ± 9	0.02975 + 0.00365	0.14174 ± 0.01131
17 Aug.	Whidby Island (Control)	96 ± 9.0	0.03150 ± 0.0037	0.1503 ± 0.0161
"	Imperial Oil	100 ± 0	0.03164 ± 0.00251	0.1582 ± 0.0125
"	Central Bedford Basin	100 ± 0	0.02944 + 0.00504	0.1472 ± 0.0252
"	Tuft's Cove	100 ± 0	0.03114 ± 0.00354	0.1557 ± 0.0177
"	Sydney Tar Pond (100%)	– **	– **	– **
"	50% Sydney Tar Pond (in control Sediment)	48 ± 46.1 *	0.00874 ± 0.00421 *	0.0224 ± 0.0258 *
"	10% Sydney Tar Pond (in control Sediment)	88 ± 17.9	0.01514 ± 0.00511 *	0.0650 ± 0.0232 *

* Significant difference (α=0.05) from control values.

** Interstitial salinity of the Sydney Tar Pond sediment sample was 4 PPT. The *Neanthes* test is not suitable this salinity.

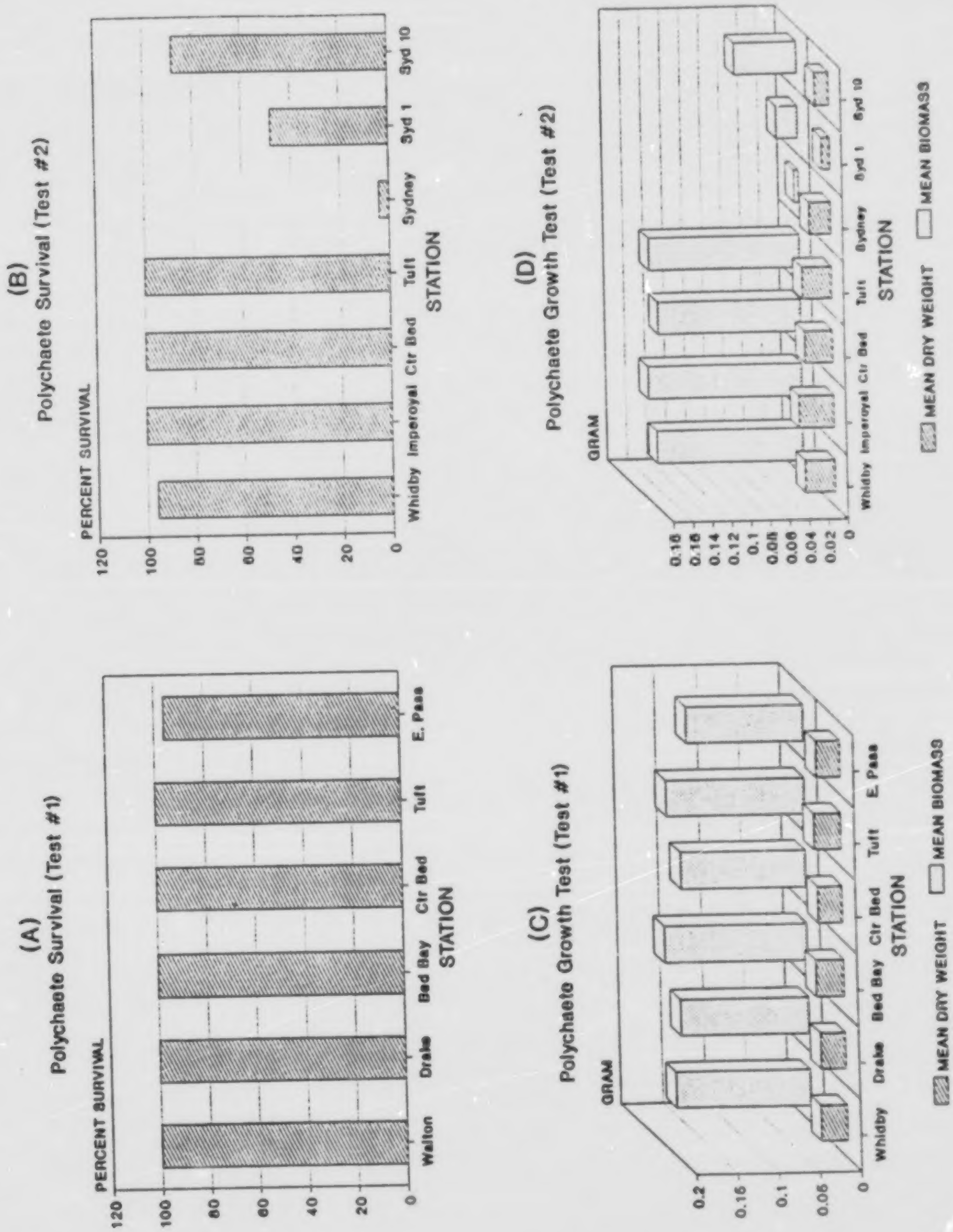


FIGURE 5. Results of polycheate bioassays in Test #1 and #2.

TABLE 7. Summary of test results for the *Macoma* bioaccumulation test on Halifax Harbour Sediment

			Contaminants in tissue's (PPM)					
Station	**Replicate**	**% Mortality (in 30 days)**	**Cd**	**Cu**	**Hg**	**Pb**	**Zn**	**PAH**
Walton	1	13.3	0.1	7.6	<0.025	9	23	0.11
	2	20.0						
Drakes Gut	1	16.7	0.1	9.2	<0.025	11	28	0.13
	2	18.3						
Tuft's Cove	1	65.0	<0.1	4.6	<0.025	12	25	ND *
	2	73.3						
Original Tissues (Not exposed to test sediments)	–	–	0.2	33	<0.025	12	23	0.01

* None detected – All 16 PAHs below detection limit.

DISCUSSION

Contaminants in Halifax Harbour Sediments

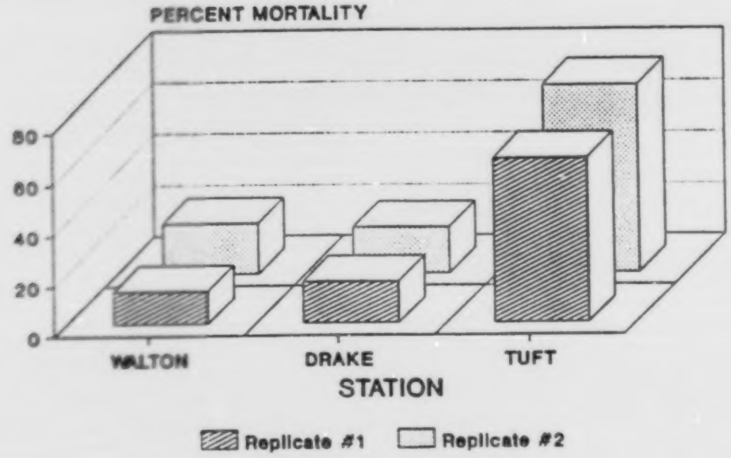
Results of the present study indicated that sediments in Halifax Harbour are enriched with heavy metals (As, Cd, Cu, Hg, Pb and Zn), organic carbon and PAH. The levels of Cu (128 ± 130.1 ppm), Pb (174.4 ± 97.6 ppm) and organic carbon (5.05 ± 1.43%) in sediments collected in the present study were similar to those reported by Buckley and Hargrave (1989). The concentrations of sediment Zn (2587 ± 4821.24 ppm) and Hg (3.58 ± 7.83 ppm) in our samples were, however, higher than previously found. The concentrations of Zn (13,584 ppm) and Hg (22.9 ppm) from the Tuft's Cove station were higher than those reported for other North American major harbours, such as Vancouver and Boston Harbours (N.O.A.A. 1988; Goyette and Boyd, 1989. Hubbard and Bellmer, 1989). Buckley and Hargrave (1989), suggested that the contamination sources of Tuft's Cove were derived from mixed sources or were dominated by a specific contaminant in a matrix of sewage waste.

Total sediment PAH concentrations (0.51 – 25.43 ppm) of this study were within the concentration range (1.34 – 36.89 ppm) reported for Vancouver Harbour (Goyette and Boyd, 1989) but much lower than those (3.25 – 2996 ppm) detected in sediments collected from 10 sites sampled within Halifax Harbour in 1987 by Environment Canada (unpublished data). Within the same sampling area located at the Tuft's Cove station, PAH concentrations of 3.32 ppm, 25.43 ppm (this study) and 2996 ppm (1987 Environment Canada unpublished data) were detected in three samples collected on different dates. These data indicated the PAH distribution in sediments from this area is highly heterogeneous.

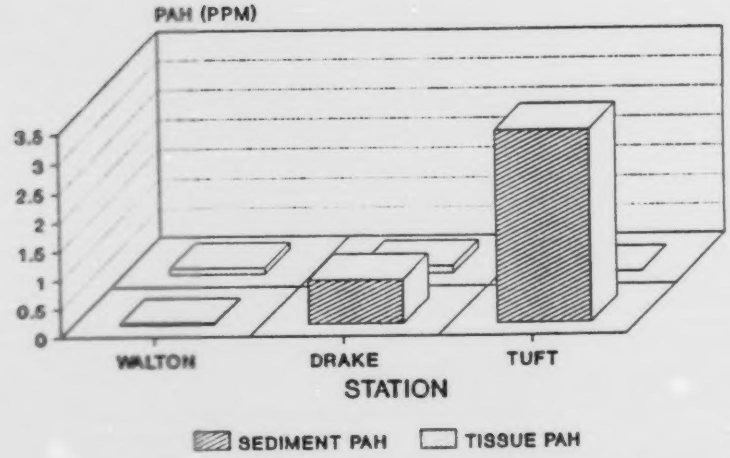
The patchiness of sediment contamination in Halifax Harbour was expected. The existence of "hot spots" is generally attributed to the integrated effects of the specific location of contaminant sources and sinks, current and sedimentation rates at the local area, degree of wave action, habitat differences of the various infaunal species and physical alterations caused by human activities.

Sediment Toxicity and Sensitivity of Test Organisms

The different biological responses expressed by the four assay organisms (Table 9) were partly due to difference in sensitivities of these organisms to the Halifax Harbour sediments. In the Microtox Test, *P. phosphoreum* was more sensitive to the solvent extract and solid phase sediments



(A)
Macoma Mortality



(B)
Sediment and Tissue PAH

FIGURE 6. (A) Percent mortality of *Macoma* in bioaccumulation test
(B) Sediment and tissue PAH concentrations in *Macoma* after 30 days exposure.

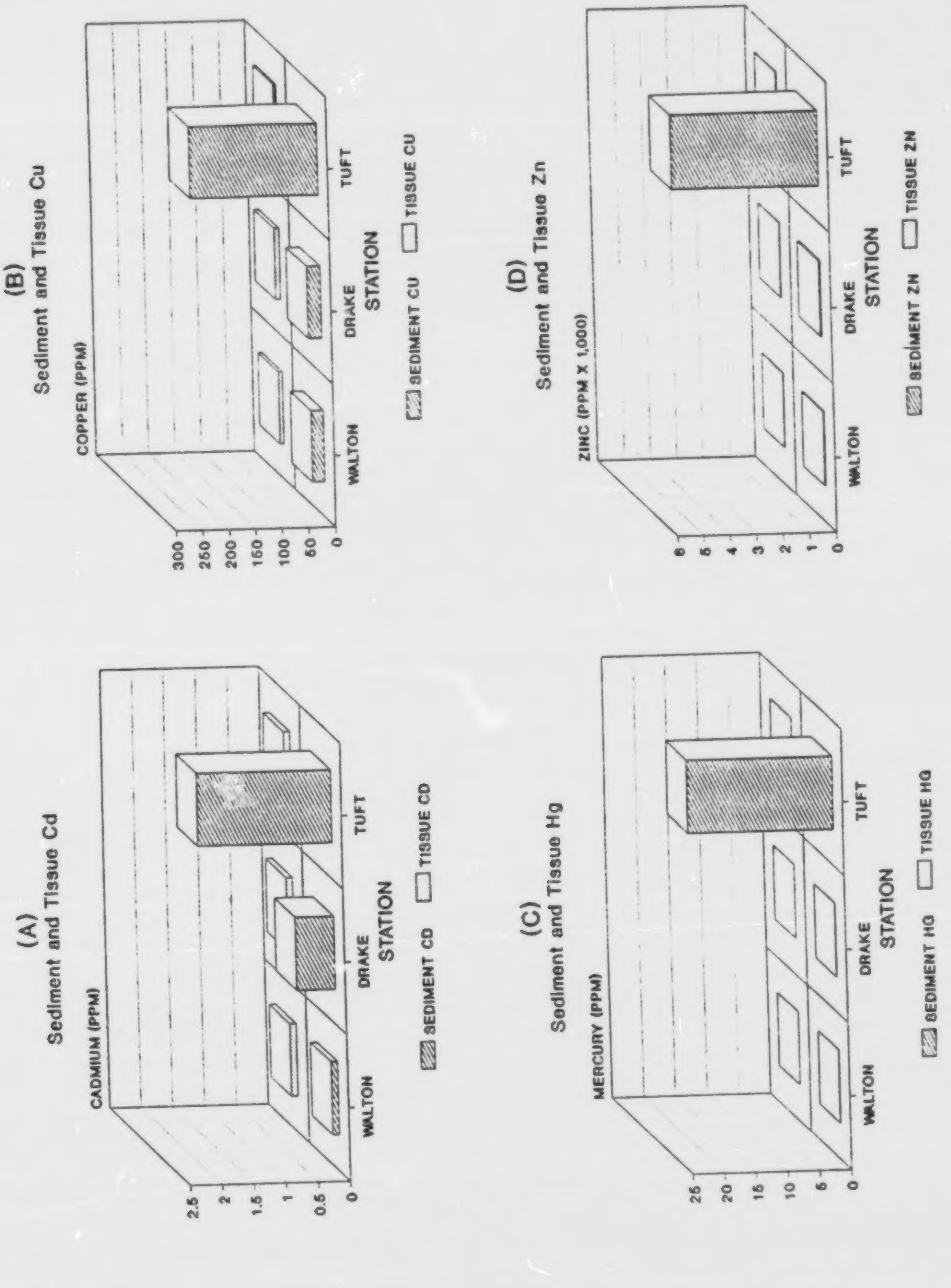


FIGURE 7. Sediment and tissue heavy metal concentrations in *Macoma* after 30 days exposure.

TABLE 8. A summary of liver lesions observed in winter flounder from Georges Bank (Control) and Halifax Harbour, N.S.

LESION	GEORGES BANK	HALIFAX HARBOUR, N.S.
Macrophage aggregation	0/9 *	3/3
Hepatic epithelial vacuolation	0/9	2/3
Hepatocyte basophilia	0/9	3/3

* Numbers of fish with lesions/number of fish examined.

than pore water. Similar results were reported by True and Heyward (1990) in a Microtox Test using interstitial water and solvent extracts obtained from Elliot Bay off Seattle, USA. They suggested that since sediment pore water contained only the water soluble contaminants, their effects on *P. phosphoreum* would only give good estimate of the toxicity of water soluble contaminants. The solvent extract sediment method gives a better estimation of the effects of particle-bound toxins. The combination of endpoints provided by the three phases of Microtox Test used in this study should give a better estimation of the toxicity of both the water soluble and particle-bound contaminants in a sediment quality assessment test. The significantly correlated results (Test #1: $r = 0.97$, $P = 0.002$; Test #2: $r = 0.99$, $P < 0.0001$) of the solid phase and solvent extract methods showed that the solid phase method is a potential tool for marine sediment toxicity assessment.

A significant correlation ($r = -0.94$, $P < 0.01$) existed between the levels of sediment PAH and the toxicity of the pore water exposure. This toxicity did not exist in sediments containing less than 9.2 ppm of PAH. The same significant associations between aromatic hydrocarbons and the results of acute toxicity tests in Microtox test have also been reported by Schiewe et al. (1985) and Giesy et al. (1988). Both the results of this present study and those reported by Giesy et al. (1988) suggested that *P. phosphoreum* is generally much less sensitive to heavy metals in the Microtox pore water test.

The relative sensitivity of *R. abronius* to sediment toxicity effects has been repeatedly demonstrated in sediment quality assessment studies on the west coast of the USA (DeWitt et al., 1988; Long et al., 1990; Robinson et al., 1988; Swartz et al., 1979, 1982; Williams et al., 1986). Williams et al. (1986) showed that results of Microtox saline extract, oyster (*Crassostrea gigas*) embryo and amphipod (*R. abronius*) bioassays were significantly correlated with one another and concurred in determination of presence or absence of toxic effects in 41% of the test sediment samples. In this study, about 50% of the results of the *R. abronius* tests concurred with the results of the Microtox solvent extract and solid phase tests. In comparison with the control and reference sediments, it is quite clear that the percent mortality of *R. abronius* in Halifax sediments were higher in sediments contaminated with high concentrations of PAH (Table 9). A concentration range of 4 ppm to 9 ppm will produce variable results while sediment concentration of > 25.43 ppm is lethal to the amphipod.

Small particle size and high organic content may sometimes cause or influence the survival of *R. abronius* exposed to uncontaminated sediments (Chapman et al., 1987; DeWitt et al., 1988; Long et al., 1990). We did not find the same effects in this study. While no *R. abronius* could survive in the coarse sediments of 10% and 50% Sydney Tar Pond sediments (2.38 – 7.36% organic carbon; 95.6 – 96.4% gravel and sand), 96% of *R. abronius* survived in the fine substrate from Tuft's Cove #1 sample (6.4% organic carbon; 76.3% silt and clay) and Eastern Passage sediments (4% organic carbon; 69.1% silt and clay).

The endpoint of the *C. volutator* survival test was less sensitive than that with *R. abronius*. We found 91% of *C. volutator* survived the Tuft's Cove sediments which was contaminated by 25.43 ppm of PAH, although only 47.3% of the survivors burrowed at the end of the test. The

Station	Total	Microtox			Amphipod		Neanthes sp.		
	PAH (PPM)	Pore Water	Solvent Extract	Solid Phase	C.v.	R.a.	Survival	Avg. dry weight	Biomass
Whidbey Island (control)	< 0.01	N	N	N	-	N	N	N	N
Walton Beach (control)	0.03	N	N	N	N	-	-	-	-
Bedford Bay	0.51	N	T	T	N	N	N	N	N
Drake's Gut	0.77	N	T	T	N	N	N	N	N
Tuft's Cove # 1	3.32	N	T	T	N	N	N	N	N
Imperial Oil	4.37	N	T	T	N	T	N	N	N
Central Bedford Basin # 2	8.49	N	T	T	N	T	N	N	N
Central Bedford Basin # 1	9.2	T	T	T	N	T	N	N	N
Eastern Passage	9.41	T	T	T	N	N	N	N	N
Tuft's Cove # 2	25.43	T	T	T	N	T	N	N	N
10% Sydney Tar Pond (in control sediment)	434.8	T	T	T	T	T	N	T	T
50% Sydney Tar Pond (in control sediment)	2538.4	-	-	-	T	T	T	T	T
Sydney Tar Pond	6615.2	T	T	T	T	NS	NS	NS	NS

C.v. = *Corophium volutator*
R.a. = *Rhepoxynius abronius*
N = Not toxic
- = Not tested
T = Toxic
NS = Test not suitable for this sediment (interstitial salinity = 4 PPT).

TABLE 9. Association between sediment toxicity and the degree of PAH contamination in sediments. Stations ranked in order based on increasing PAH concentration.

different sensitivities of these two species of amphipod might be due to their different living habitats *C. volutator* is a tube dwelling amphipod (Bousfield, 1973). In comparison with *R. abronius* which is a subsurface burrower, *C. volutator* is frequently exposed to the overlying water as water is pumped through the tube. The limited animal-sediment contact thus reduces the chances of direct exposures of this animal to the particle-bound contaminants in the sediment.

The endpoint of the juvenile *Neanthes sp.* bioassay was less sensitive than that of the Microtox and *R. abronius* bioassays. Correlation Coefficient Analysis indicated a possible negative relation between the concentrations of sediment PAH and the mean biomass of surviving polychaete ($r = 0.76$, $P = 0.08$, $n=6$). The lack of sensitivity of this tube dwelling species to contaminated sediments in this study is quite different from the results reported by Johns et al. (1990). Further evaluation is needed to assess the usefulness of this animal in sediment quality tests.

Bioavailability of Sediment Contaminants in Halifax Harbour Sediments

The uptake experiment in this study indicated that after 30 days of continuous exposure to the Tuft's Cove #1 sediments from Halifax Harbour, no increased concentrations of tissue contaminants were detected in *Macoma*. This result confirms the suggestion in Buckley and Hargrave (1989) that, since most of the organic-rich sediments from Halifax Harbour are in a state of strong chemical reduction, the high concentrations of metals such as Zn, Pb and Hg are probably present as highly insoluble metal sulfides.

The lack of contaminant uptake by *Macoma* in our study might also be due to the following factors: (1) the 30 day exposure time was probably too short for any significant uptake of contaminants in the test animals. Studies conducted by other workers in both laboratory and field experiments showed that uptake of heavy metals (Samant et al., 1990) and PAH (Malins et al. 1982) occurred after various species of clams were exposed to contaminated sediments for more than 70 days; (2) relatively high levels of organic carbon in the Tuft's Cove sediments might reduce the uptake of contaminants (Landrum et al., 1987; Swartz et al., 1990; Di Toro et al., 1990); (3) toxic effects of the sediment may have reduced contaminant uptake (e.g. by interfering with feeding).

Winter Flounder Histopathological Studies

Epizootics of hepatic neoplasia in bottom-feeding fish have been described in many areas contaminated by municipal and industrial effluent (Harshbarger and Clark, 1990). In the most comprehensively studied case, that of English sole (*Parophrys vetulus*) in Puget Sound, there is a strong statistical association between the presence of contaminants, especially PAH, and the development of neoplasia (Malins et al., 1985). These and other contaminants are also suspected to be important in other epizootics of teleost neoplasia.

The results of the present study are preliminary, given the small sample size, but they do suggest that histological changes are evident in winter flounder collected from the Halifax Harbour. The changes seen in the liver of these flounder were mild, but comparable to those reported from Boston Harbour, and other contaminated New England waters. The hepatic changes observed included three of the lesions commonly seen from the Boston Harbour flounder, namely epithial cell vacuolation, macrophage aggregation and hepatocyte basophilia. Additionally perisinusoidal edema was observed in one fish. Epithelial cell vacuolation is a very common lesion in flounder from Boston Harbour, and other heavily contaminated harbours. It is closely associated with neoplastic change, although the precise role of these cells is currently debated (Moore, In prep.). Macrophages aggregate under a number of influences, including chemical contamination (Blazer et al., 1987). Hepatocyte basophilia is commonly seen in fish from contaminated waters (Moore, In prep.). It reflects a loss of lipid and glycogen stores, and an increase in endoplasmic reticulum and other nucleic acid containing organelles. Perisinusoidal edema is a rare lesion also seen in fish from Boston, and in flounder experimentally exposed to dietary benzo(a)pyrene (Moore, In prep.).

Seminomas are not common. Cases registered at the Registry of Tumours for Lower Animals include *Zebra danio*, African lungfish, and winter flounder (Harshbarger per. comm.). In a review of fish tumour epizootics (Harshbarger and Clark, 1990) it is suggested that gonadal neoplasms occur in a pattern unrelated to environmental pollution.

Gill epithelial hyperplasia has been described from other contaminated areas. Flounder from Quincy Bay, Boston Harbour, were shown to have partial or complete occlusion of the inter – lamellar space (Gardner and Pruell, 1988). The lesions observed in the Halifax Harbour flounder are primarily limited to basal proliferation. The same basal proliferation was observed in winter flounder fed dietary estradiol (Moore, In prep.). The link between these observations is unclear, but it is interesting that many of the organochlorine contaminants found in coastal sediments are estrogenic (Nelson et al., 1978), and that estradiol induces respiratory epithelial proliferation in rodents.

CONCLUSION

The results of the present study showed detectable biological effects of the Halifax Harbour sediments on two (*P. phosphoreum* and *R. abronius*) of the four species of benthic organisms tested. Mortality in *Macoma* was increased, although no significant uptake of contaminants were detected in the bioaccumulation test. The hepatic lesions observed in winter flounder collected from the harbour indicated that long term chronic effects of the harbour contaminants were detectable.

The Microtox Test was the most sensitive bioassay for sediment toxicity assessment. The test was most effective for screening the presence of toxic compounds in the sediment when the three phases of test were used concurrently. *R. abronius* was relatively sensitive to the Halifax sediments. For sediments contaminated with intermediate levels of pollutants, the results of this amphipod bioassay were variable. However, because of the relatively high concordance between the responses of this amphipod test and the Microtox test, it was most effective in serving as a supporting study for the Microtox Bioassay. Based on their sensitivities to sediment contaminants observed in this study, we ranked the bioassays as follows: Microtox Solvent Extract > Microtox Solid Phase > *R. abronius* Bioassay > Microtox Pore Water > *C. volutator* and *Neanthes sp.* Bioassays.

For regulatory purpose, such as the assessment of sediment quality for dredged spoils ocean dumping permit applications, where cost effectiveness and efficiency of test are essential for decision making, only the three Microtox Tests and the *R. abronius* are useful in assessing the degree of contamination at the regulatory levels of contaminants. There is, therefore, a recognized need for continuing research and, development of toxicity bioassays in regulatory programs such as the Ocean Dumping Program under the Regulations of the Canadian Environmental Protection Act (CEPA). Tests such as the Microtox and amphipod bioassays should be further evaluated with other existing methods, such as the Microbial Enzyme Bioassay (Tay, 1989), Oyster Embryo Bioassay (Williams et al., 1986), Sea Urchin Embryo Bioassay (Long et al., 1990), to compare their precision, reliabilities, sensitivities, discriminatory power and ease of use.

The presence of lesions in the few winter flounder examined indicated that, in addition to the lethal bioassays and bioaccumulation studies, chronic effect measurements of sediment contaminants is essential for any pollution control programs such as the Halifax Harbour Clean Up Project. Without collection of baseline data and biological response studies, it is impossible to monitor the results of any pollution control programs which are designed to clean up the marine environment.

The results of this study are preliminary. Because of the limited data set and sampling size for proper statistical analysis and the small number of fish used for histological observation, there should not be any attempt to extrapolate our results and conclusions should not be accepted uncritically.

ACKNOWLEDGEMENTS

We thank D. Aggett and W. Barchard of the Marine Environment Branch, Environment Canada, Atlantic Region, for their support in providing funds and technical advice to many parts of this project. Special thanks also to B. Hargrave and D. Peer Of the Bedford Oceanography Institute, Fisheries and Oceans Canada, for their cooperation in planning and carrying out the field portions of this program; to G. Atkinson of Environment Canada, Ottawa, for his assistance in statistical analysis; to A. McIver of Environment Canada, Atlantic region, for his field collection of the Sydney Tar Pond sediment. We express our special appreciation to O. Vaidya, M. MacIsaac, P. Hennigar and B. MacDonald of the Environment Canada Regional Laboratory for conducting the chemical analyses. D. Casavechia of the Halifax Fisheries Laboratory provided the winter flounder. C. McPherson and D. Taylor-Prime of the Environment Canada Regional Library provided invaluable literature support.

REFERENCES

Blazer, V.S., Wolke, R.E., Brown, J. and C. A. Powell. 1987. Piscine macrophage aggregate parameters as health monitors: effect of age, sex, relative weight, season and site quality in largemouth bass (*Micropterus salmoides*). Aquatic Toxicology 10: 199-215.

Bousfield, E.L. 1973. Shallow-water Gammaridean Amphipoda of New England. Cornell University Press, Ithaca, N.Y. 312 pp.

Buckley, D. E. and B. T. Hargrave 1989. Geochemical characteristics of surface sediments. In: H.B. Nicholls (Ed.) Investigations of marine environmental quality in Halifax Harbour. Canadian Technical Report of Fisheries and Aquatic Sciences #1693. Fisheries and Oceans Canada. pp 9-36.

Chapman, P. M., R. N. Dexter, R. M. Kocan and E. R Long. 1984. An overview of biological effects testing in Puget Sound, Washington: Methods, results, and implications. In R. D. Cardwell, R. Purdy and R. C. Bahner (Ed.) Aquatic Toxicology and Hazard Assessment (7th Symposium). STP 854. American Society for Testing and Materials, Philadelphia, PA. pp. 344-362.

Chapman, P. M., R. N. Dexter and E. R. Long. 1987. Synoptic measures of sediment contamination, toxicity, and infaunal community composition (the Sediment Quality Triad) in San Francisco Bay. Mar. Ecol. Prog. Ser. 37: 75-96.

DeWitt, T. H., G. R. Ditsworth and R. C. Swartz. 1988. Effects of natural sediment features on survival of the Phoxocephalid amphipod, *Rhepoxynius abronius*. Mar. Environ. Res. 25: 99-124

Di Toro, D. M., J. D. Mahony, D. J. Hansen, K. J. Scott, M. B. Hicks, S. M. Mayr and M. S. Redmond. 1990. Toxicity of cadmium in sediments: The role of acid volatile sulfide. (submitted to Environ. Tox. Chem.)

Environment Canada. 1989. Ten-day test for sediment toxicity using the marine infaunal amphipod, *Rhepoxynius abronius*. Environment Canada Aquatic Biological Test Method, Environmental Protection, Environment Canada (First Draft, March 1989).

Gardner, G. R. and R. J. Pruell. 1988. A histopathological and chemical assessment of winter flounder, lobster, and soft-shelled clam indigenous to Quincy Bay, Boston Harbour, and *in situ* evaluation of oysters including sediment (surface and cores) chemistry. US EPA. 76pp.

Giesy, J. P., R. L. Graney, J. L. Newsted, C. J. Rosiu, A. Benda, R. G. Kreis, Jr. and F. J. Horvath. 1988. Comparison of three sediment bioassay methods using Detroit River sediments. Environ. Toxicol. Chem. 7: 483-498.

Goyette, D. and J. Boyd. 1989. Distribution and environmental impact of selected benthic contaminants in Vancouver Harbour, British Columbia, 1985 to 1987. Environment Canada Regional Program Report:89-02. 99 pp.

Harshbarger, J. C.and J. B. Clark.1990. Epizootiology of neoplasms in body fish of North America. Sci. Tot. Environ. 94: 1-32.

Hubbard, W. A. and R. J. Bellmer. 1989. Biological and chemical composition of Boston Harbour, USA. Mar. Pollut. Bull. 20(12): 615-621.

Johns, D. M., T. C. Ginn and D. J. Reish. 1990. Protocol for juvenile *Neanthes* sediment bioassay. PTI Environmental Services, Bellevue, Washington, 98007, USA. 13 pp.

Landrum, P. F., S. R. Nihart, B. J. Eadie and L. R. Herche. 1987. Reduction in bioavailability of organic contaminants to the Amphipod *Pontoporeia hoyi* by dissolved organic matter of sediment interstitial waters. Environ. Tox. Chem. 6: 11-20.

Long, E. R., M. F. Buchman, S. M. Bay, R. J Breteler, R. S. Carr, P. M. Chapman, J. E. Hose, A. L. Lissner, J. Scott and D. A. Wolfe. 1990. Comparative evaluation of five toxicity tests with sediments from San Francisco Bay and Tomales Bay, California. Environ. Tox. Chem. 9: 1193-1214.

Luna, L. G. 1968. Manual of histologic staining methods of the Armed Forces Institute of Pathology. McGraw-Hill. 253 pp.

Malins, D. C., B. B. McCain, D. W. Brown, A. K. Sparks, H. O. Hodgins and Sin-Lam Chan. 1982. Chemical contaminants and abnormalities in fish and invertebrates from Puget Sound. N.O.A.A. Technical Memorandum OMPA-19. 168 pp.

Malins, D. C., M. M. Krahn, D. W. Brown, L. M. Modes, M. S. Myers, B. M. McCain and Sin-Lam Chan. 1985. Toxic chemicals in marine sediment and biota from Mukilteo, Washington: Relationships with hepatic neoplasms and other hepatic lesions in English Sole (*Parophrys vetulus*) J. Nat. Can. Inst. 74: 487-494.

McKeague, J. A. (Ed). 1978. Manual on soil sampling and methods of analysis. In: Canadian Society of Soil Science, 1978. Section 3.8.

Microbics. 1989a. Microtox data collection system installation/information. Version 2. Microbics Corporation, 2232 Rutherford Road, Carlsbad, California, 92008 USA.

Microbics. 1989b. A Microtox manual: How to run toxicity tests using the Microtox Model 500. Microbics Corporation, 2232 Rutherford Road, Carlsbad, California, 92008 USA.

Nelson, J. A., R. F. Struck and R. James.1978. Estrogenic activities of chlorinated hydrocarbons. J. Tox. and Environ. Health. 4: 325-339.

Nicholls, H. G. (Ed). 1989. Investigations of Marine Environmental Quality in Halifax Harbour. Canadian Technical Report of Fisheries and Aquatic Sciences. 1693. Fisheries and Oceans Canada. 83 pp.

N.O.A.A. 1988. A summary of selected data on chemical contaminants in sediments collected during 1984 – 1989. N.O.A.A. Tech. Mem. #OMA44.

Ribo, J. M. and K. L. E. Kaiser. 1987. *Photobacterium phosphoreum* Toxicity Bioassay. I. Test procedures and applications. Tox. Assess. 2: 305-323.

Ribo, J. M. and F. Rogers. 1990. Toxicity of mixtures of aquatic contaminants using the Luminescent Bacteria Bioassay. Tox. Assess. 5: 135-152.

Robinson, A. M., J. O. Lamberson, F. A. Cole and R. C. Swartz. 1988. Effects of culture conditions on the sensitivity of a phoxocephalid amphipod, *Rhepoxynius abronius*, to cadmium in sediment. Environ. Tox. Chem. 7: 953-959.

Samant, H. S., K. G. Doe and O. C. Vaidya. 1990. An integrated chemical and biological study of the bioavailability of metals in sediments from two contaminated harbours in New Brunswick, Canada. The Science of the Tot. Environ. 96: 253-268.

Schiewe, M. H., E. G. Hawk, D. I. Actor and M. M. Krahn. 1985. Use of a bacterial bioluminescence assay to assay toxicity of contaminated marine sediments. Can. J. Fish. Aquat. Sci. 42: 1244-1248.

Swartz, R. C., W. A. DeBen and F. A. Cole. 1979. A bioassay for the toxicity of sediment to the marine macrobenthos. J. Water Pollut. Contr. Fed. 51: 944-950.

Swartz, R. C., W. A. DeBen, K. A. Sercu and J. O. Lamberson. 1982. Sediment toxicity and the distribution of amphipods in Commencement Bay, Washington, USA. Mar. Pollut. Bull. 13(10): 359-364.

Swartz, R. C., W. A. DeBen, J. K. P. Jones, J. O. Lamberson and F. A. Cole. 1985. Phoxocephalid amphipod bioassay for marine sediment toxicity. In: R. D. Cardwell, R. Purdy and R. Comotto-Bahner (Ed) Aquatic Toxicology and Hazard Assessment: Seventh Symposium, ASTM. STP 854. American Society for Testing and Materials, Philadelphia, PA.pp.284-307.

Swartz, R. C., F. A. Cole, D. W. Schults and W. A. DeBen. 1986. Ecological changes in the Southern California Bight near a large sewage outfall: Benthic conditions in 1980 and 1983. Mar. Ecol. Prog. Ser. 31: 1-13.

Swartz, R. C., P. F. Kemp, D. W. Schults, G. R. Ditsworth and R. J. Ozretich. 1989. Acute toxicity of sediment from Eagle Harbour, Washington, to the infaunal amphipod, *Rhepoxynius abronius*. Environ. Tox. Chem. 8: 215-222.

Swartz, R. C., D. W. Schults, T. H. DeWitt, G. R. Ditsworth and J. O. Lamberson. 1990. Toxicity of fluoranthene in sediment to marine amphipods: A test of the equilibrium partitioning approach to sediment quality criteria. Environ. Tox. Chem. 9: 1071-1080.

Tay, K. L. 1989. The role and application of environmental bioassay techniques in support of the impact assessment and decision-making under the Ocean Dumping Control Act in Canada. Hydrobiologia 188/189: 595-600.

True, C. J. and A. A. Heyward. 1990. Relationships between Microtox Test results, Extraction Methods, and physical and chemical compositions of marine sediment samples. Tox. Assess. 5: 29-45.

Walton, A. 1978. (Ed) Methods for sampling and analysis of marine sediments and dredged materials. Ocean Dumping Report #1. Environment Canada. 74 pp.

Williams, L. G., P. M. Chapman and T. C. Ginn. 1986. A comparative evaluation of marine sediment toxicity using bacterial luminescence, oyster embryo and amphipod sediment bioassays. Mar. Environ. Res. 19: 225-249.

LINKING ENVIRONMENTAL INFORMATION TO DECISION-MAKING: THE ROLE OF MONITORING, REFERENCE THRESHOLDS AND INDICATORS

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Paper presented at the Marine Environmental Quality Monitoring Platform Session of the 17th Annual Aquatic Toxicity Workshop, November 5-7, 1990, Vancouver, Canada

ABSTRACT

To manage ecosystems and their resources on a sustainable basis, sound information must be available on which decisions can be based and evaluated. Environmental reporting – the timely delivery of understandable environmental information – is an important tool which can help provide this information to decision-makers and the public. In recognition of this, Environment Canada is leading a federal effort to develop a national profile of environmental indicators.

This paper examines how environmental indicators can be used to focus environmental information and facilitate its use for decision-making and public communication. The indicator concept is introduced, and the basic building blocks required for indicator development – such as environmental monitoring and reference thresholds – are discussed. Examples of indicators which have been used successfully are presented and analyzed. The ability of existing data sets to allow for the use and development of marine environment indicators is reviewed and found wanting. Suggestions are made from an environmental indicator viewpoint regarding the development of a national marine monitoring program. The need for data which permit temporal and spatial analysis at national and regional scales is emphasized.

INTRODUCTION – THE NEED FOR ENVIRONMENTAL INFORMATION

We live in a period of heightened awareness and concern about the state of the environment. The issues which are fuelling public and scientific concern range from those which are global in nature with the potential to affect all of humanity (such as stratospheric ozone depletion) to those which are regional or of importance in a specific location (such as poor water quality). This concern will intensify as human populations and environmental pressures expand and as more is learned about the nature and significance of environmental problems.

In Canada, as in many other countries, the public is demanding more information about the state of the environment, about how it is changing over time, and about the ecological, economic and health-related significance of current conditions and trends. Equally important is the demand for actions to resolve existing environmental problems and for strategies which anticipate and prevent potential and emerging problems.

These demands are increasing pressures on decision-makers at all levels. In 1987, the World Commission on Environment and Development served urgent notice that "based on the latest and best scientific evidence, decisions must be taken now to ensure sustainable human progress and human survival" (World Commission on Environment and Development, 1987). Canadian marine managers are under pressure as illustrated by present concerns regarding, for example, the state of the fishery and the presence of highly toxic dioxin and furan contaminants in shellfish, seabirds and marine mammals.

It goes without saying that sound decisions require sound information. Canadians as decision-makers want to receive information about the state of the environment in – the same systematic way they receive information about the state of the economy (Government of Canada, 1990). In the marine environment, intelligent decisions regarding the use and allocation of resources require consistent information about the status and trend of environmental quality in coastal areas. Key questions include whether environmental conditions are stable, improving or deteriorating, how conditions in different areas compare with one another and which specific areas require special attention, research and remedial action.

Environmental reporting – the timely delivery of understandable environmental information – is an important tool which can provide environmental information to decision-makers and the public. This paper will examine how environmental reporting, through the use of indicators, can facilitate this process. The indicator concept will be introduced and the basic building blocks required for indicator development – such as environmental monitoring and reference thresholds – will be reviewed. Examples of currently used environmental indicators will be presented and their essential features explained. Finally, the ability of current marine monitoring programs to meet environmental reporting needs will be examined and suggestions made regarding the collection of additional information.

DEFINING AND CHARACTERIZING ENVIRONMENTAL INDICATORS

Over the past twenty years (ie. since the Stockholm Conference) governments have increased the resources aimed at environmental protection and conservation. The growth of non-government action has been at least as significant. However, many unanswered questions remain. One important group of questions relates to how we will know whether or not progress is being made. Economic indicators such as the Gross National Product, the Consumer Price Index and the national current account reveal key information about economic performance. In contrast, there are few systematically reported indicators which inform us of trends in the state of the environment and on which past management decisions can be evaluated.

Interest in sustainable development and growing public concern about environmental pressures have stimulated governments to re-examine their capacity to assess and monitor the state of the environment and detect changing conditions and trends. Pressures are also growing for measures of performance, thus the subject of environmental indicators has come to attract considerable attention as a necessary tool for helping to chart progress being made toward a sustainable future (Organisation for Economic Cooperation and Development, 1990).

This interest is reflected in several recent initiatives. At the international level, in May 1989, the OECD Council meeting at Ministerial Level called, *inter alia*, for a next generation work programme on environmental economics that would integrate environment and economic decision-making more systematically and effectively as a means of contributing to sustainable development. In July 1989 the Paris G-7 Economic Summit (the annual summit meeting of the world's seven leading industrial democracies) reinforced this and called on the OECD "within the context of its work on integrating environment and economic decision-making, to examine how selected environmental indicators could be developed". Here in Canada, Environment Canada is leading a federal effort to develop a national prototype set of environmental indicators based on existing data.

Indicators Defined

A review of the literature reveals that the term "indicator" has achieved widespread use in many disciplines, including environmental reporting. Perhaps due to this widespread use, the term's meaning appears to have expanded to the point where it is commonly used as a synonym for data (Gelinas and Slaats, 1989). In the context of this paper, an indicator is a statistic or measure which facilitates interpretation and judgement about the condition of an element of the world or society in relation to a standard or goal (modified from U.S. EPA, 1972).

Environmental indicators are selected key measures which reveal or summarize some aspect of the state of the environment, natural resource assets and related human activities. They focus on measures of environmental change and convey how the environment is responding to both stresses on the environment and management responses (Kerr, 1990).

Characteristics of Environmental Indicators

Criteria for selecting environmental indicators have been reviewed by several analysts (Gelinas and Slaats, 1989; Ward, 1990 among others). These are summarized below and have implications for the design of data collection and monitoring programs.

Policy Relevance

From a decision-making viewpoint, policy relevance is an essential feature of an environmental indicator. The utility of an indicator is a function of the extent to which it informs us of our progress in moving toward (or away from) established goals, targets or objectives (discussed in detail in the section on Reference Thresholds).

Cause/effect is a second factor related to policy relevance. Indicators which summarize information about environmental conditions or stresses where a cause-effect relationship has been established are less "ambiguous" than those where such a relationship is weak or poorly understood.

Detection of Temporal and/or Spatial Trends

As noted above, decision-makers, in deciding where to focus and allocate resources, require information on whether conditions are stable, improving or deteriorating and on how conditions in different areas compare with one another.

A common problem is with deciding on where resources should be allocated and mitigating strategies applied.

By presenting information on environmental conditions and pressures using data which have been collected consistently over time and space, indicators can provide needed guidance to decision-makers.

Geographic Scope

There is a spatial continuum in which environmental indicators can be applied which ranges from the site-specific to the global level. However, the extent of geographic coverage will often influence the utility of an indicator, which generally increases with size of spatial coverage. For issues of national significance an indicator should provide a national picture of the phenomenon being reported.

Scientifically Rigorous

Environmental indicators should be technically sound and their attributed significance should be scientifically defensible. There should be general consensus among credible experts regarding the validity of the indicator.

Understandable

Indicators should be easily understood by Canadians and by decision-makers. What the indicator represents and the significance of the values reported should be understandable. In this regard, the use of reference thresholds is especially helpful in communicating indicators and interpreting their significance.

ESSENTIAL BUILDING BLOCKS: ENVIRONMENTAL MONITORING AND REFERENCE THRESHOLDS

Environmental indicators cannot be developed and applied in the absence of key building blocks; specifically, the availability of rigorous and consistent data, and thresholds against which to interpret reported values. The strategic importance of these elements is depicted in Figure 1, and is reviewed below.

Environmental Monitoring: Role and Design Considerations

Environmental monitoring can be defined as the repetitive observing of one or more environmental parameters according to pre-arranged schedules in space and time, using comparable methodologies for environmental sensing and data collection (O'Neil, 1990). The role of monitoring, simply stated, is the collection of data and information. As such, it is a key element in any environmental management or decision-making process. The data generated through monitoring can provide a sound scientific basis for management decisions. Monitoring can provide a record of changes occurring in key environmental indicators and help explain why such changes are occurring. (The relationship between marine monitoring and decision-making is reviewed in detail by the National Research Council, 1990.)

However, to maximize the utility and value-added of environmental monitoring, and to allow for the development of environmental indicators, several factors (among others) should be kept in mind (reviewed in detail by Philips and Segar, 1986; Wolfe, 1987; National Research Council, 1990). First, the management objectives and information needs that monitoring is to address should be clearly identified. In such a management-oriented approach to monitoring, social, economic and environmental values should be represented as directly as possible in the selection of parameters to be monitored (Wolfe, 1987). As a basis for making and priorizing decisions, monitoring information should inform us about whether important objectives are being met and uses sustained.

Second, the data generated should be comparable over time and space. Without such consistency temporal trends in environmental conditions cannot be identified and spatial comparisons cannot be made. This requires that the sampling protocols and analytical procedures inherent to the monitoring program be consistent.

A third and key consideration relates to the scope of monitoring. Is monitoring to.be carried out on an integrated ecosystem-wide basis or is it to be more restricted in scope by focusing on selected issues and concerns (such as contaminant presence)? This is a key question which must be addressed. The integrated approach involves ecosystem-wide monitoring where all elements of an ecosystem are surveyed for change in condition.

From a coastal pollution assessment perspective, the monitoring of contaminant fate and effect merits special attention. The presence and concentration of contaminants

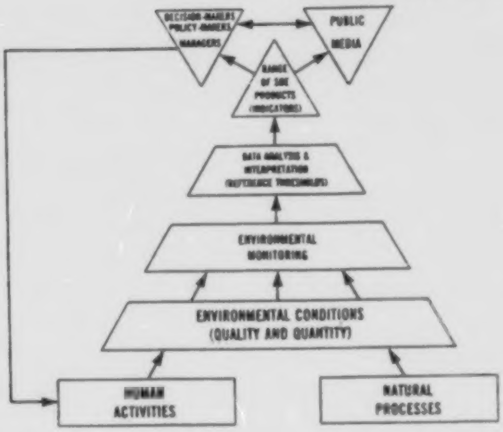

FIGURE 1. The Relationship Between Monitoring, Reporting and Decision-making (modified from Kerr, 1990)

an be measured in three media: water, sediments and in living organisms. The extent of contamination, and the potential effects, cannot be gauged without recognizing that these three components act as a single system. Rather than choosing just one medium to monitor, information from all three should be incorporated to obtain an integrated view of contaminant presence and effect in the environment (Wolfe, 1987).

Monitoring for contaminant presence also has implications for monitoring program design. For decision-making and environmental reporting purposes, it is important that the data generated through monitoring reflect conditions at different contamination gradients. For investigations of temporal and spatial variability in contamination, a number of sites ranging from highly contaminated to background would be chosen, probably in inshore locations. In contrast, detection of long-term trends in environmental quality would require the monitoring of sites further offshore, removed from the influence of important contaminant sources (Philips and Segar, 1986). The "gradient to background" approach to monitoring was endorsed by a recent Environment Canada – Memorial University workshop on environmental effects monitoring (Environment Canada and Memorial University, 1988). Also important is that the monitoring sites be chosen so that they reflect the broad picture regarding the state of the environment. This is a problem given the statistical limitations of sampling design. Insufficient knowledge regarding the extent of contamination and impact introduces an element of chance into monitoring design which must survey and infer broader conditions from information collected at fixed locations. Shalski (1990) reviews this problem and suggests a method of rotational surveying which increases the statistical confidence associated with monitoring results.

Objectives and Reference Thresholds: Essential Interpretive Tools

Environmental indicators show whether or not objectives are being met, or progress made vis-a-vis identified goals. This approach implies that objectives be clearly identified prior to the identification of indicators. Objectives, in the sense meant here, are broad goals that policy-makers are striving to meet and that are generally deemed desirable to attain. One model of such objectives is presented in Figure 2.

Reference thresholds associate closely with objectives and aid with interpretation of data. Interpretation of data is an essential feature, and one of the truly value-added aspects of environmental reporting (Kerr, 1990). Raw data are meaningless to decision-makers and the public until summarized and interpreted, therefore methods of aggregating and presenting data over time and space, and of interpretation, are needed. Reference thresholds offer one way of interpreting data in a meaningful way and, like objectives, are essential to indicator development. Environmental quality is a concept largely based on human values. Environmental objectives articulate these values while reference thresholds quantify them by identifying ranges of desirable/undesirable conditions or activities (eg. normal, problem, critical and irreversible) (Gelinas and Slaats, 1989). Environmental indicators juxtapose monitoring data against threshold levels of environmental quality and thus

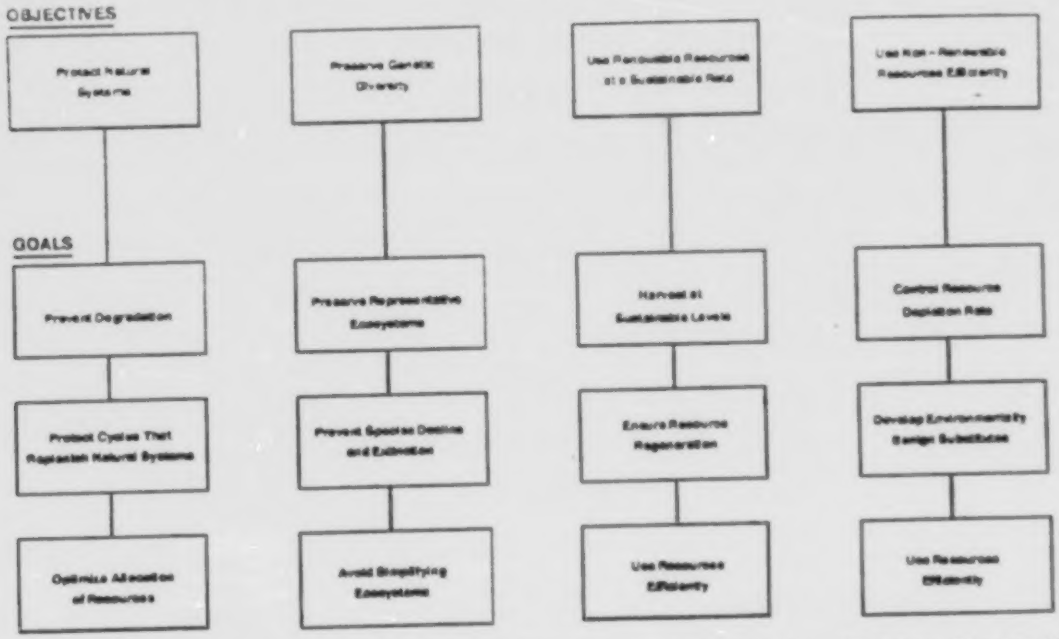
OBJECTIVES
Protect Natural Systems
Preserve Genetic Diversity
Use Renewable Resources at a Sustainable Rate
Use Non-Renewable Resources Efficiently
GOALS
Prevent Degradation
Preserve Representative Ecosystems
Harvest at Sustainable Levels
Control Resource Depletion Rate
Protect Cycles That Replenish Natural Systems
Prevent Species Decline and Extinction
Ensure Resource Regeneration
Develop Environmentally Benign Substitutes
Optimize Allocation of Resources
Avoid Simplifying Ecosystems
Use Resources Efficiently
Use Resources Efficiently

FIGURE 2. Schematic Presentation of Sustainable Development Objectives (modified from Sheehy, 1989)

facilitate assessment of the state of particular conditions (ie. whether conditions are acceptable or unacceptable), and of whether they are improving or deteriorating (ie. moving toward or away from threshold levels of environmental quality).

As denoted here, the term "reference threshold" is loosely used to denote any threshold against which monitoring data can be compared. The use of reference thresholds is portrayed in Table 1; examples of threshold "types" are depicted in Table 2.

USING ENVIRONMENTAL INDICATORS: ILLUSTRATIVE EXAMPLES

Environmental indicators can be grouped into two categories: single-measure indicators and composite indicators (sometimes referred to as indices). Most of what are commonly referred to as environmental indicators fall into the former category while many of the commonly reported economic indicators fall into the latter.

To illustrate the concepts outlined in previous sections of this paper two examples of indicators are presented: 1) acid deposition as an indicator of acidification stress on the environment; and 2) the Ontario Air Quality Index (AQI) as an indicator of environmental quality. These have been selected as representative of the two indicator types mentioned above (single measure, composite) and because they make effective use of monitoring data and reference thresholds to influence decision-making and public awareness. As such, each indicator is examined from the following viewpoints:

- relation to a societal goal;
- monitoring and data support, including spatial and temporal coverage;
- use of reference thresholds;
- policy relevance and utility to decision-making.

Example Indicator – Acid deposition

Acidification of the environment is a regional problem which has affected much of the central and eastern Canadian environment. Environmental indicators have been used to assess and portray the extent of the problem and to track progress achieved in resolving it.

TABLE 1. Example of the Relationship Between Objectives, Reference Thresholds and Indicators

Objective:	Prevent Environmental Degradation
Reference Threshold:	Standard for effluent toxicity
Indicator:	Measure of performance in meeting standard (indicator identifies whether or not effluent meets established toxicity threshold)

TABLE 2. Examples of Reference Thresholds

Type of Threshold	Application
Environmental quality guideline, objective or standard	Contaminant levels in environmental media or effluents (e.g. water quality guidelines, Apparent Effects Thresholds)
Activity target or goal	Target established as a policy goal (eg. DFO "no net loss" of habitat, Montreal Protocol Targets for phaseout of CFCs)
Comparative threshold	Data compared against background conditions or conditions at other reference sites.
Ecological Threshold	Contaminant levels which induce adverse biological effects (e.g. eggshell thinning); minimum population size required for viability.

Societal goals

Resolution of the acid rain problem speaks to two broad goals has outlined in Figure 2):

- Protection of natural systems and prevention of degradation;
- Preservation of biological diversity and prevention of species extinction and decline.

Monitoring and Data Support

This indicator is constructed from and supported by an extensive data collection and management system – the Canadian Air and Precipitation Monitoring Network (CAPMON). An array of parameters associated with the acid rain issue are monitored daily at 24 stations in rural areas from all provinces and territories save Yukon and P.E.I.. Monitoring is site specific at regionally representative sites. The indicator can be presented for specific sites or summed and portrayed over larger areas. Sampling and analytical procedures are intercalibrated and the system allows for detection of spatial and temporal deposition trends.

Use of Reference Thresholds

Reference thresholds have been developed which allow the data collected from CAPMON to be communicated and used through an indicator. Through scientific research a threshold level ranging between and 15 kg per hectare of wet sulphate deposition has been identified (exact amount is a function of local area characteristics). Deposition above this level will likely result in damage to aquatic and terrestrial ecosystems (e.g. reduced biodiversity and productivity).

Policy Relevance and Use for Decision-making

Policy-makers have made extensive use of this indicator. Related indicators such as aquatic pH and loss of biodiversity signalled in the 1970s that acidification was a problem requiring urgent attention. Establishment of a threshold level of sulphate deposition and identification of the gap between current and desirable conditions led to the formulation of an emission reduction program designed to decrease sulphate deposition loadings into the environment. By tracking the wet sulphate deposition indicator, progress made in attaining stated goals can be monitored and the success of recent acid rain control policies evaluated.

Example Indicator – Ontario Air Quality Index

An index reduces complex and voluminous monitoring data for an array of parameters into a single number which can be used to give an indication of overall environmental quality (Federal - Provincial Committee on Air Pollution, 1980). There have been few quantitative indices developed for environmental quality only those related to air quality have achieved widespread use in Canada (some countries, like the United Kingdom and New Zealand, have developed water quality indices). The Ontario Air Quality Index (AQI) is a good example of such an index. The index is used as a tool to better inform the public on the quality of Ontario's air and reports this quality in the following unit-free numerical classifications.

0 - 15	very good
16 - 31	good
32 - 49	moderate
59 - 100	poor
>100	very poor

Societal Goals

The goals which the index relate to are the protection of human health from air pollution and the protection of natural systems by minimizing air pollution effects on the environment.

Monitoring and Data Support

The data used to derive the AQI are obtained from a monitoring network consisting of 33 monitoring sites in 26 cities across the province. The system provides hourly information for six air pollutants (sulphur dioxide, suspended particles, nitrogen dioxide, ozone, carbon monoxide and total reduced sulphur compounds). Air quality samples are collected and analyzed in a consistent manner allowing for spatial and temporal comparisons.

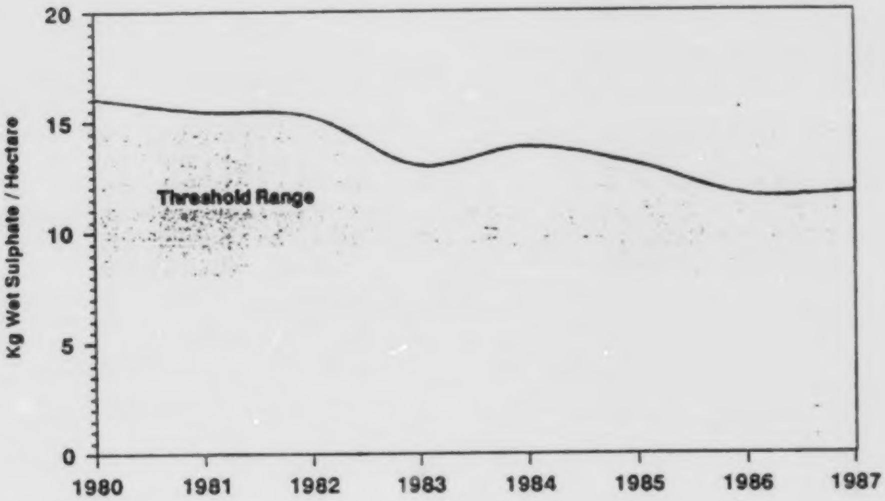

FIGURE 3. Example Indicator – Trends in Wet Sulphate Deposition: Eastern Canada (M. Still, Pers. comm.)

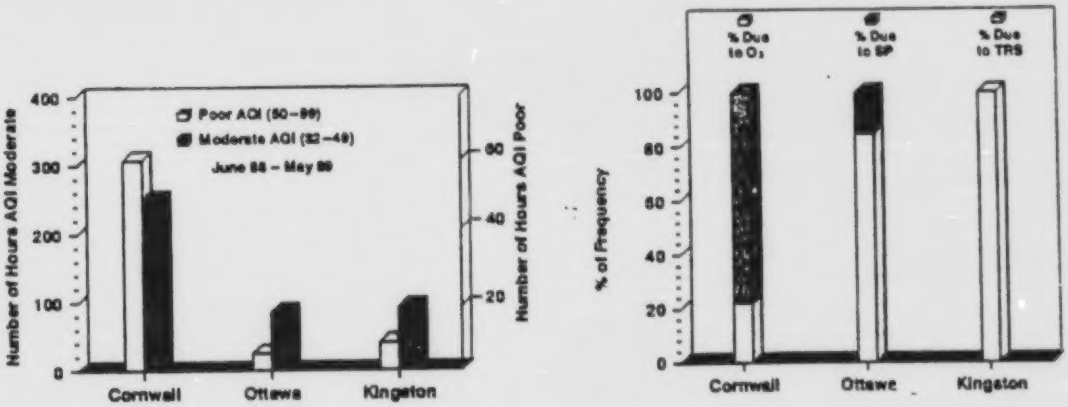

FIGURE 4. Ontario Air Quality Index (Yap et al., 1989)

Use of Reference Thresholds

Reference thresholds are essential to the development of the index and have been established for each of the six pollutants in the index (Federal and Ontario Air Quality Standards). The hourly concentration of each of the pollutants is calibrated to a common sub-index scale ranging from zero upwards (the same scale as the "quality" categories presented above). The sub-index is calculated for each pollutant based on its effect and the highest sub-index at any time becomes the value of the overall AQI.

Policy Relevance and Use in Decision-Making

The AQI is one of the most widely reported environmental quality indicators. The AQI levels for each city are reported several times daily to the news media and there is much interest from the media and the public when the air quality is in the Moderate and Poor ranges.

From a decision-making viewpoint, the index can, and does, report the number of times each pollutant is responsible for values of the index falling in the moderate to poor ranges of air quality. Armed with this information, policy-makers can direct resources toward lowering levels of the most critical air pollutants. Decisions can be taken to temporarily close air pollution sources such as power plants if air quality conditions deteriorate to critical levels.

CRITIQUE OF EXISTING DATA AND IMPLICATIONS FOR MARINE ENVIRONMENTAL QUALITY MONITORING

A 1985 program review of major surveys in Canada found that many were costly and inefficient. It criticised the overall system of environment and natural resources monitoring as lacking a clear focus and integration of their activities. The review concluded that "in those survey departments having a scientific research sector, the national policies and objectives for the science are not always clearly articulated to management" (Canada Task Force on Program Review, 1985).

Generally, many of our environmental monitoring programs were developed and implemented years ago when the very nature of environmental issues and management needs was substantially different. In the past, concerns focused on relatively simple environmental problems and monitoring was largely oriented to single issues and single-media surveys. The complexity of today's issues requires a more comprehensive understanding of pressures on the environment and environmental change (Kerr, 1990).

This paper reviews how monitoring, reference thresholds and indicators can be used to effectively develop and apply environmental information for decision-making purposes. But what form of marine environmental monitoring is required to meet today's marine management needs? For environmental reporting, with its focus on cross-sectoral analysis, and for marine management, an ecosystem approach to monitoring is recommended.

Marine ecosystems are dynamic entities which provide an array of living and non-living resources, support a variety of (often conflicting) uses and have tremendous ecological and intrinsic value. Comprehensive ecosystem monitoring, in the marine context, would involve monitoring of human activities with the potential to adversely affect marine ecosystems (e.g. contaminant inputs, fishing), components of the marine environment at varying levels of organisation (e.g. coastal habitats, populations of marine species) and the concentration and effects of contaminants and pollutants (Table 3).

TABLE 3. Elements of Integrated Marine Monitoring

A. Pressures on the Marine Environment

- contaminant inputs
- coastal land-use
- harvesting activity

B. Environmental Condition/State

- concentrations of contaminants (water, sediments, biota)
- Biotic state (biomass and population change, species health, bio-diversity)
- ecosystem change (eg. productivity)

The desired scope of marine monitoring is thus an important question to consider. Other considerations include the adequacy of existing Canadian marine monitoring relative to management needs, and whether the data generated by such programs are nationally consistent and comparable. A better knowledge of what is being monitored in the marine environment, where, and by whom is needed. These questions, among others, should be addressed as part of the larger process of developing a national marine environmental monitoring program in Canada.

Recent Canadian reviews of marine environmental quality (Kay, 1989; Environment Canada, 1989) concluded that existing data allowed for a snapshot assessment of the state of the marine environment but could not support any firm conclusions regarding whether or not conditions were stable, improving or deteriorating. Requests by international organisations like the Organization for Economic Co-operation and Development (OECD) for national-level data on marine environmental quality cannot be adequately met. The problems with existing marine data can be summed as follows:

- data often result from limited monitoring at specific sites. Such monitoring may well meet the needs it was intended for, however it often precludes temporal analysis of trends and spatial comparison among regions;
- sampling and analytical protocols often differ, even within departments or agencies. Thus even if two groups are monitoring similar parameters results may not be comparable;
- there are gaps in the coverage of monitoring. A full analysis of existing monitoring relative to identified management needs would determine more precisely critical data gaps however, experiences to date point to gaps in, among other things, pollutant inputs (an input "budget" cannot be developed for the Canadian marine environment from existing data), contaminant distribution and effect and compliance with environmental standards and regulations.

SUMMARY OF KEY CONSIDERATIONS

Environmental reporting in Canada is still in an early stage of development. Many of the monitoring programs and data sets upon which current reporting efforts rest were designed to respond to site-specific and sector-specific requirements and issues. Much work remains to be done to identify the indicators which should be tracked, to put in place the monitoring and data support systems which will allow this to be done and to develop the reference thresholds which will facilitate interpretation of data. This is true of the marine environment as well as of other environmental sectors.

As work proceeds towards the elaboration of a marine monitoring programme, the following steps should be addressed:

1. National and regional marine management objectives and issues for which information is required should be clearly identified.
2. Reference thresholds which identify desired levels of environmental quality should be developed. To the extent possible, these should be quantitative and linked with management objectives.
3. The information required to identify whether thresholds are being achieved and objectives met should be determined. Existing monitoring efforts which meet objectives should be identified and evaluated, and gaps pin-pointed. Monitoring programs required to address information gaps should be designed and implemented. Consideration should be given to ensuring national and regional spatial and temporal consistency.
4. Management information is produced only when it is delivered to managers and decision-makers in a usable, accessible form (National Research Council, 1990). Using indicators, information on trends in marine environmental quality and on areas and issues of concern should be reported in an understandable and systematic way to decision-makers and the public.

REFERENCES

Canada Task Force on Program Review. 1985. Major Surveys: A Study Team Report to the Task Force on Program Review. Ottawa.

Environment Canada. 1989. Marine Environmental Quality in Canada Status and Priorities. Draft of April 1989, prepared by the Marine Environmental Quality Advisory Group.

Environment Canada and Memorial University of Newfoundland. 1988. Point Source Effects Monitoring for the Marine Environment – Bridging the Gap Between Theory and Practice. Report on Seminar/ Workshop of March 15 and 16, St. John's, Newfoundland

Gelinas, R. and J. Slaats. 1989. Selecting Indicators for State of the Environment Reporting. Technical Report Series Report No. 8 (draft). Strategies and Scientific

Methods Division, State of the Environment Reporting Branch, Environment Canada, Ottawa

Government of Canada. 1990. A Framework for Discussion on the Environment. Ministry of Supplies and Services, Canada

Kay, B. H. 1989. Pollutants in British Columbia's Marine Environment: A Status Report. SOE Report No. 89-1, Environment Canada

Kerr, M. A. 1990. From Monitoring to Decision-Making: The Application of State of the Environment Reporting. In: Proceedings of the International Conference and Workshop "Global Natural Resource Monitoring and Assessments: Preparing for the 21st Century", Volume 2, American Society for Photogrammetry and Remote Sensing: 741-752

National Research Council. 1990. Managing Troubled Waters – The Role of Marine Environmental Monitoring. National Academy Press, Washington, D.C.

O'Neil, D. 1990. A DOE Management Framework, Strategy and Action Plan for Environmental Monitoring. Draft of August 27, 1990. Environment Canada.

Organisation for Economic Co-operation and Development. 1990. Environmental Indicators Draft Progress Report. Environment Directorate Document ENV/EC/SE(90)27. Paris, France

Philips, D. J. I. and D. A. Segar. 1986. Use of Bioindicators in Monitoring Conservative Contaminants: Program Design Imperatives. Mar. Poll. Bul. 17(1): 10-17

Shalski, J. R. 1990. A Design for Long-Term Status and Trends Monitoring. Journal of Environmental Management. 30:139-144

Sheehy, G. 1989. The Use of Indicators to Show the State and Recent Trends in Sustainable Development: A Forestry Pilot Study. Technical Report Series Report No. 11, Strategies and Scientific Methods Division, State of the Environment Reporting Branch, Environment Canada, Ottawa

Still, M. (Atmospheric Environment Service, Environment Canada). Personal communication.

United States Environmental Protection Agency. 1972. Quality of Life Indicators. Environmental Studies Division, Office of Research and Monitoring.

Ward, J. C. 1990. Environmental Indicators for State of the Environment Reporting. Information Paper No. 21. Centre for Resource Management, Lincoln Centre and University of Canterbury, New Zealand.

Wolfe, D. A. 1987. Marine Pollution Monitoring: Objectives and Design Criteria. In Ocean 87 Proceedings, W. McNab & Son, Halifax, N.S., pp 1532-1537

Worf, D. C. (ed). 1989. Biological Monitoring for Environmental Effects. D.C. Health and Co., United States

World Commission on Environment and Development. 1987. Our Common Future. Oxford University Press. Oxford and New York.

Yap, D.; D. Mignacca and D. Fraser. 1989. A Year Experience With the Air Quality Index System for Ontario. Paper presented at the Technology Transfer Conference. Toronto, Canada

MONITORING BACTERIOLOGICAL POLLUTION IN BRITISH COLUMBIA SHELLFISH GROWING WATERS

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ABSTRACT

Monitoring programs to assess the sanitary quality of bivalve molluscan shellfish growing waters have been conducted on a regular basis by Environment Canada since the early 1970's. These programs measure bacteriological contamination in water, sediments and biota and also identify and evaluate sources of pollution to harvesting and growing areas through detailed shoreline investigations. Surveys are conducted according to international protocols and water quality is assessed against a national water quality standard. This paper reviews survey design protocols, including sampling strategies and statistical considerations, and presents trend data in pollution levels during the past 20 years. New data management techniques are discussed, including the implementation of GIS mapping, and observations are made concerning the application of some of the monitoring principles used in the shellfish program to other marine environmental quality assessment studies.

INTRODUCTION

The positive relationship between sewage pollution of shellfish growing areas and enteric disease has been well documented (NSSP, 1989; Hunt 1977), and a recent summary of shellfish-related disease outbreaks in the United States shows an increasing incidence of illness associated with unknown agents (Table 1). In Canada it has been estimated that almost 2 million cases of foodborne illness occur annually, representing an economic loss of approximately $2.2 billion (Farber, 1989). Reports of shellfish-borne disease outbreaks in Canada as summarised by Rippey (1989) are not up to date but are shown in Table 2.

Shellfish-borne infectious diseases are generally transmitted by the faecal-oral route, the cycle usually begun with faecal contamination of the shellfish growing waters. Faeces, either through the disposal of human sewage by marine outfalls or via landwash from urban (eg. malfunctioning septic fields) or agricultural (animal faecal matter, manure etc.) areas can release pathogenic bacteria and viruses into the marine environment.

Comprehensive monitoring programs to assess the sanitary quality of shellfish harvest areas were initiated in the United States following an outbreak of typhoid fever in the winter of 1924-25. This outbreak involved 1500 cases and 150 deaths and was traced to contaminated oysters. Canadian programs were instituted soon thereafter.

Monitoring Program Requirements

Monitoring programs to assess the sanitary quality of shellfish growing waters are designed to assess the degree and extent of contamination, identify the sources of pollution and provide predictions of pollutant transport and/or incidences. Techniques include intensive bacteriological monitoring surveys, hydrographic and dye release studies, outfall modelling and shoreline investigations and plant evaluations.

Water Quality Standard

Epidemiological investigations have had difficulty in establishing a direct numerical correlation between the bacteriological water quality and the degree of hazard to human health. Investigations of typhoid and other enteric disease outbreaks in the United States between 1917 and 1925 established that diseases would not normally be attributed to shellfish when the growing waters had total coliform levels of less than 70/100 mℓ, provided the areas were not exposed to direct contamination from fresh sewage. Further studies in the 1970's established a faecal coliform value of 14/100 mℓ as being equivalent to the 70/100 mℓ total coliform level.

In Canada, shellfish growing areas may be designated as "Approved" if the median or geometric mean faecal coliform Most Probable Number (MPN) does not exceed 14/100 mℓ and not more than 10% of the samples exceed a faecal coliform MPN of 43/100 mℓ under adverse pollution conditions.

Bacteriological Studies

Bacteriological sampling programs are designed to assess the degree and extent of faecal pollution at times which would reflect the most adverse pollution conditions. Adverse pollution conditions may occur on a seasonal basis (e.g. rainfall impacts on landwash and sewage treatment plant operation or summer anchorage and/or seasonal use areas), or even a daily basis (eg. tidal impacts, variable discharge rates from sewage plants etc.). When an area has never been sampled, 5 – 15 samples are collected from each sampling station to obtain baseline data. Following this initial survey, a minimum of 5 samples are collected annually at key sample sites to evaluate water quality and these data are re-evaluated every three years to ensure the growing waters are properly classified.

Shoreline Surveys

Shoreline surveys are conducted to identify all actual and potential sources of pollution to the shellfish growing area. These studies may include evaluation of the operation of sewage treatment plants, door-to-door interviews of residents, application of Best Management Practices at farms, etc. These surveys are conducted concurrently with the bacteriological sampling program to better define the impacts on the marine environment.

Hydrographic Surveys

Evaluation of the hydrographic characteristics of a shellfish area may often be necessary to determine the time of transport of pollutants and dilution/dispersion characteristics. Current patterns, tidal influences, flushing rates, stream/river flows may require evaluation using a number of techniques including dye releases, droque studies and river gauging.

Data Interpretation

Proper evaluation of the sanitary quality of shellfish growing areas requires the integration of a number of information sources and survey activities as noted above. Sampling strategies may often be altered during the survey in response to data being generated from the shoreline or hydrographic studies. It is also important to recognise that the methods used in the collection and analysis of bacteriological data will affect the interpretation of results. Data collected under varying pollution conditions should not be combined for analysis as this may result in an improper classification of the growing area. The lack of precision in the MPN test procedure should also be recognised in evaluating the data. As Woodward (1957) points out,the 95% confidence limits for a five tube MPN test cover a thirteen-fold range from approximately 24% to 324% of the MPN. The precision of MPN estimates can be improved by increasing the frequency of sampling under similar conditions. Generally, confidence in the median or geometric mean is rapidly lost with fewer than 10 samples, while collection of 20 or more samples will not significantly improve confidence levels (Furfari, 1979).

B.C. SHELLFISH PROGRAM

Sampling Program

In British Columbia, shellfish growing water quality is evaluated through a sampling network comprised of approximately 2,400 marine and 1,000 freshwater sampling sites, from which 8,000 bacteriological samples are collected annually. Data from these sampling sites is used to assess the adequacy of shellfish closure boundaries, evaluate water quality at new aquaculture and harvest sites and quantify pollution levels in point and non-point sources. Since 1972 approximately 2,200 km of the 13,750 km of coastline on Vancouver Island and the southern mainland has been surveyed. The surveyed coastline includes most of the 12,400 hectares of shellfish production area (clam beaches and aquaculture leases) identified by the Department of Fisheries and Oceans and provincial Ministry of Crown Lands (pers. comm.).

Closure Statistics

There are currently 140 sanitary shellfish closures in B.C. encompassing 71,000 hectares and 760 km of coastline. Figure 1 shows closure trends from 1972 – 1990 according to pollution source type. Not all areas under closure are productive shellfish habitat (eg. Vancouver Harbour, Victoria coastline) however closures are in place for public health protection purposes. The greatest closed area results from multiple pollution sources, followed by sewage outfalls, agricultural/hinterland drainage, boat sewage discharges and urban runoff. Closures from agricultural/-hinterland drainage have shown the most dramatic increase since 1986, principally as a result of studies in previously unsurveyed areas on the west coast of Vancouver Island. Although the data shows an increasing closure trend, it is not possible to attribute this to increasing pollution since baseline data does not exist in many of the areas more recently surveyed.

Data Management

The Shellfish Program data management system presently under development combines a PC platform GIS mapping system, QUIKMap, with an Oracle database on the regional

FIGURE 1. Trends in B.C. Shellfish closures 1972 - 1979.

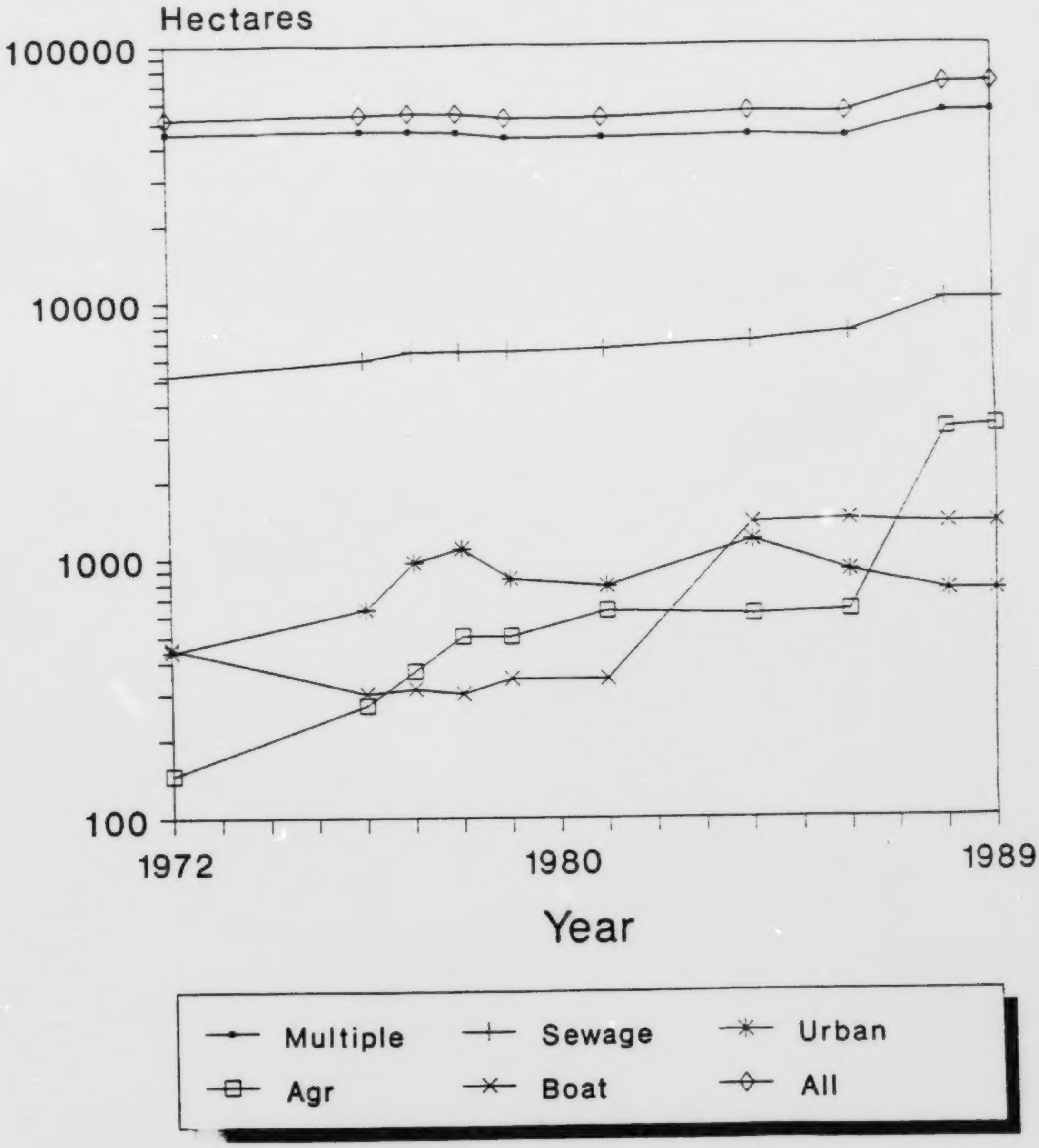

TABLE 2. Shellfish-borne disease outbreaks in Canada 1974–1980

Year	Disease or etiological agent	No. of cases
1974	Unknown	68
1975	"	101
1976	"	63
1977	"	63
1978	"	57
1979	"	74
1980	"	144

VAX computer. Currently most of the southern coast, including all of Vancouver Island, has been digitised at a scale of 1:40,000 from Canadian Hydrographic Service charts. Georeferenced databases include sampling stations, provincial discharge permits, aquaculture leases, clam beaches, and shellfish closures (as polygons).

SUMMARY

The design of monitoring programs to assess the sanitary quality of shellfish growing waters has some unique considerations as compared to other environmental quality measurement programs including:

- biased sampling (adverse pollution conditions)
- concurrent pollution source identification and evaluation
- does not measure ecosystem effects
- changing monitoring strategies based on daily review of data/information

These "design criteria" have been developed from the perspective of measuring a public health threat rather than assessing environmental effects. The measurement of bacteriological contamination in the marine environment also reflects immediate pollution events (either continuous or transient) rather than cumulative contamination.

Future sampling strategies will include programs to measure other contaminants in the shellfish (mussel watch programs) to obtain baseline contaminant levels, identify "hot spots" and assist in regulatory action. A preliminary evaluation of Mussel Watch programs has been undertaken (Sojo et al, 1990) and further discussions on development of a national program are ongoing.

REFERENCES

Farber, J. M. 1989. Foodborne Pathogenic Microorganisms: Characteristics of the Organisms and Their Associated Diseases I. Bacteria. Can. Inst. Food Sci. Technol. J. 22(4):311.

Furfari, S. A. 1979. Bacteriological Data Analysis. In: Shellfish Growing Area Survey and Evaluation Training Course, Lexington, Mass., May 17-21, 1982.

Hunt, D. A. 1977. Indicators of Quality for Shellfish Waters. In "Bacterial Indicators/Health Hazards Associated With Water." A. W. Hoadley, B. J. Dutka eds. ASTM Special Technical Publication 635.

NSSP, 1989. National Shellfish Sanitation Program Manual of Operations. Part I. Sanitation of Shellfish Growing Areas. 1989 Revision. U.S. Department of Health and Human Services. Public Health Service. Food and Drug Administration.

Rippey, S. R. 1989. Shellfish Borne Disease Outbreaks. Shellfish Sanitation Program Technical Report. Department of Health and Human Services. Public Health Service. Food and Drug Administration.

Sojo, L. E., M. C. Hamilton, D. Files. 1990. National Mussel Watch Program. Phase I. Overview of Existing Mussel Watch Programs. Draft Report prepared for Environment Canada by Seakem Oceanography Ltd.

Woodward, R. L. 1957. How Probable is the Most Probable Number. Journal AWWA 449:1060.

RIVERINE INPUTS IN MARINE ENVIRONMENTAL QUALITY MONITORING

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The contribution of rivers is an important part of marine environmental quality monitoring because rivers may represent an important source of contamination to the marine environment. Riverine input is especially important in coastal areas with restricted circulation and large river flows such as some bays and estuaries.

The riverine flux of a contaminant may be calculated as the product of the riverine concentration of the contaminant and the river water discharge. To understand the true riverine flux of a contaminant, one must know the actual values of both the contaminant concentration and the river flow and understand how each is affected by anthropogenic and natural processes.

In the past, there have been some large programs to measure the quality of many rivers, and some of the data from these programs is being used to project trends in stream quality and flutes of contaminants to the coastal zone. But how reliable are these data? Marine chemists have recognized for years that contamination is a problem in the sampling, handling and analysis of environmental samples, especially the ultra-trace concentrations found in open ocean waters. Since there has been a lack of standard reference materials, marine chemists have had to rely upon intercalibration exercises to assess the quality of their methods. A series of intercalibrations of dissolved metals in seawater have resulted in vastly improved results for the analysis of trace metals in seawater in recent years.

The United States Geological Survey has been collecting a large database of dissolved trace metal concentrations in rivers as part of the National Stream Quality Assurance Network (NASQAN). During a study of global river geochemistry, we sampled most of the large rivers along the east coast of the US and Canada, along with some smaller rivers. Some of these sampling sites were near sampling stations that are included in the NASQAN network. Thus, we were able to compare our data with those in the NASQAN database for the same sampling period.

We used clean sampling and handling techniques during periods of high and low flow and averaged the data to represent a mean concentration. Our data for Cu, Cd, Pb and Zn are consistently lower than those reported in the NASQAN database, Cd, Pb and Zn being 1 to 2 orders of magnitude lower (Windom, et al., 1990). Our results are similar to those obtained by other investigators (Schiller and Boyle, 1985; 1987), which leads us to believe that the NASQAN data are not suitable in determining riverine flutes or trends in trace metal concentrations.

It is also necessary to assure that data being reported by different laboratories in an environmental quality monitoring program are comparable. The best method for determining this is by intercalibration exercises. Even with the higher concentrations usually found in rivers, it is necessary to perform these types of exercises.

Results from a recent intercalibration of marine sediments that was performed among laboratories in Asia and the United States, show that even for elements that are relatively easy to sample and analyze, such as iron and manganese, there can be large systematic variations among labs.

If we are assured that the data represent true concentrations, are these concentrations the result of natural or anthropogenic processes? Our results for East Coast rivers suggest that although dissolved cadmium and zinc appear to be enriched in these rivers, these elements vary systematically with pH. The same holds for dissolved lead, although the lead concentrations in some rivers may reflect a depletion of lead in the soils of the drainage basin due to extensive weathering. The conclusion is that the dissolved concentrations of these trace metals may be influenced more by the chemistry of the rivers than by anthropogenic sources.

Given the particle reactive nature of many trace metals (and synthetic organic compounds) and the effects of river chemistry upon dissolved concentrations, the concentrations of trace metals on suspended particles may be a better method of assessing environmental quality than dissolved trace metal concentrations. If we calculate an enrichment factor for trace metals on suspended particles in East Coast rivers versus average crustal material, we see that enrichments are best seen on particles from rivers having small suspended loads. In rivers with large concentrations of suspended material, the enrichment factor is low and relatively constant, reflecting dilution of contaminants by natural material.

If we trust the data to represent the true trace metal concentrations of a sample, can we use the concentrations with discharge data to calculate the real flux? The concentrations of riverborne materials often vary with the discharge. This can often be estimated by a rating curve, an equation of the form $C = aQ^b$ where C is the concentration, Q is the discharge, and a and b are constants. These rating curves may vary dramatically among rivers (GESAMP, 1987, and references therein), so that contaminant concentrations may be greatly affected by changes in river flow.

Storm related events have a large impact upon the transport of material by rivers and may account for the majority of the annual riverine flux of a substance. The concentrations of some constituents vary with discharge, some reflect dilution by the larger water flow, while some increase through the storm period, reflecting a flushing effect of material stored in the drainage basin. Depending upon the intervals between storm events, antecedent effects from previous storm events may reduce the amount of transport in successive storm events.

It is necessary to understand the magnitude of riverine inputs in any study of coastal marine environmental quality. One must be assured that both past and present riverine data is both reliable and comparable. Natural processes, including temporal changes in riverine fluxes, must be understood to discern variations in environmental quality that are due to anthropogenic influences.

REFERENCES

GESAMP - IMO/FAO/UNESCO/WMO/WHO/IAEA/UN/-UNEP Joint Group of Experts on the Scientific Aspects of Marine Pollution. 1987. Land/Sea Boundary Flux of Contaminants: Contributions from Rivers. Rep. Stud. GESAMP, 32.

Schiller, A. M. and E. Boyle. 1985. Dissolved zinc in rivers. Nature. 317:49-52.

Schiller, A. M. and E. Boyle. 1987. Variability of dissolved trace metals in the Mississippi River. Geochem. Cosmochem. Acta. 51:3273-3277.

Windom, H. L., J. T. Byrd, R. G. Smith, Jr. and F. Huan. 1990. Can historical dissolved trace metal data be used to assess metal trends in the Nation's rivers? (submitted).

STATISTICAL CONSTRAINTS OF STATUS AND TRENDS ANALYSIS OF ENVIRONMENTAL QUALITY DATA: TRENDS IN CONTAMINANT LEVELS IN FISH

EXTENDED SUMMARY

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INTRODUCTION

Investigation of time trends in contaminant levels in fish stocks has been of primary interest to the International Council for the Exploration of the Sea (ICES) since 1974. Conventional analysis of variance (ANOVA) and analysis of covariance (ANCOVA) have been widely used for analyzing data on contaminant levels (ANOVA) or contaminant levels with length as the covariate (both transformed to logarithms) (ANCOVA), measured on individuals from several years of collection. ANCOVA is an inadequate procedure when regression slopes (coefficients of regression of contaminant concentration on length) for separate years and residual variances about these separate regressions are not equal for all years. By employing a weighted procedure (Misra et al. 1990) this methodology is extended to analyze such data sets where homogeneity of regression coefficients and residual variances is not present.

It must be further recognized that:

1. The scope of a univariate analysis is restricted because it analyses data on only one contaminant at a time. Thus, a series of univariate ANCOVAs will ignore the mutually correlated structure of the dependent (Y_i) variables. Multivariate analysis of covariance (MANCOVA) will not ignore correlations among Y_i variables as data on all contaminants are analyzed jointly. Information about relationships, interdependence, and relative importance of variables is retrievable from the MANCOVA. The ANCOVA model is fully absorbed in the MANCOVA model.

2. Frequently, analysis of real data has shown the need for more than one covariate to be employed, e.g. for approximately one-half of 62 contaminant trend data sets collected under the Cooperative Monitoring Programme of the ICES, statistically improved analyses were achieved when more than one biological covariate (length) was employed. Year-means must be adjusted for variations in all available biological covariables. Biological covariables are generally mutually correlated. In particular, biological characters such as length, weight, age, are highly correlated as they are all related to fish size. Although several techniques are available for handling this problem of multicollinearity, no technique works well in all circumstances or is without constraint. The approach we employ to overcome the problem of multicollinearity (Misra et al. 1989) uses principal components (PCs) of biological covariables within the MANCOVA of Canadian Atlantic cod data – the PC-MLR method.

WEIGHTED ANALYSIS OF COVARIANCE

The weighted procedure presented here assumes that only one covariate is in use. The procedure can be extended to the case where two or more covariates are employed. The general problem in many circumstances is to examine how a quantity is varying on a medium to long-term scale when its concentration (Y) is correlated with an associated variable (X), the influence of which is not fully controllable by sampling. Its concentration is, in any case, likely to vary between years as other, associated parameters vary. Comparison of multiple (K) means is potentially difficult in any case. Our study was undertaken to find an acceptable way of analyzing data sets that are not amenable to the conventional ANCOVA. Several procedures for multiple comparison of means exist (e.g. see Miller 1981). There is not complete agreement on how best to conduct multiple comparisons (Harris 1975) or even on the general principles of multiple comparisons (Miller 1981).

Also, poor data structure, a characteristic of many monitoring programs (ICES 1989), would induce irregularities due to improper definition of the effects of the covariate on contaminant concentration levels. Consistency of a regression coefficient depends not only on the sample size but also on the widths of ranges of the covariate for individual years. Data with wide ranges of the covariate have the following additional advantage (Li 1965): Points with covariate values near their mean $(\bar{X})$ will not necessarily prevent the regression line from rotating. On the other hand, points with large deviations from the mean and thus located towards either end of a wide range will tend to stabilize the line, as these points are much more effective in preventing the line from rotating. Sometimes, for practical reasons, close adherence to the sampling guidelines may be impossible but its importance should never be underestimated. Homogeneity of regression coefficients among years is still not assured as further causes of variation (other than the sampling procedure) in the values of the estimate, b_j, of regression coefficients also exist. For example, the ANCOVA model of the ICES guidelines employs only one covariate (X). It was also felt (ICES 1986b) that ignoring other covariables is not prudent, especially when some studies have shown the necessity of taking their effect into account. The ICES model with several covariates and the model with only one covariate, length, were fitted to 53 CMP data sets and their fits compared based upon their residual standard deviation values (ICES 1987b; Nicholson and Wilson 1987). The model with a single covariate gave increased residual standard deviations in about half the cases. In two cases, the increase exceeded 20% (ICES 1987b). In a similar analysis of contaminant data from Atlantic cod (*Gadus morhua*), Misra et al. (1989) reported increases in residual standard deviations of up to 107% for contaminant concentration data and up to 328% for contaminant burden (contaminant concentration times organ weight) data.

Biological variables available as covariates, e.g. length, weight, and age are frequently mutually correlated as they are related to fish size/growth (ICES 1989, 1987a). Omission of other covariates, which are correlated with the covariate X, employed in the regression, will bias the value of b_j. (Draper and Smith 1981, Bliss 1970). It is further noted from Draper and Smith (1981, Chapter 6) that screening of the correct suite of covariates is not an easy task. Sometimes more than one "best" suite can be identified. Several techniques are available for this purpose and yet no technique works well in all circumstances, or is always better than all others. The Working Group on Statistical Aspects of Trend Monitoring (ICES 1989) noted that when covariates are correlated, the probability of selecting the covariates by a stepwise regression approach (which is considered to be one of the best of the variable selection procedures; Draper and Smith 1981) is low. Second and higher order associations (interactions) of omitted covariates alter value(s) of the coefficient(s) of regression on the covariate(s) retained in the regression model (Bliss 1970). The value observed when a biological character is measured on an individual may be referred to as the "phenotypic value" of that individual (Falconer 1972).

The above observations concerning statistical correlations between biological covariates relate to their phenotypic correlations (Falconer 1972). Further probing into the possible causes of correlation between variables requires extension of the phenotypic correlations to genetic and environmental correlations and to the examination of several possible causes of variations in these (Falconer 1972). A good sampling strategy, which includes a prudent choice of the study area, e.g. choosing a stable stock as opposed to an erratic or widely spread population, will help to stabilize the correlation between variables from one year to the another but variances and covariances of biological characteristics are also properties of populations and of the environmental circumstances to which individuals are subjected (Falconer 1972). In actual studies the occurrence of heterogeneity of statistical parameters among years is, therefore, to be expected. At this time the true mathematical model incorporating varying b_j values, if there is one, is unknown. In the real world, therefore, we can expect to encounter data sets where β_j values are not the same for all years. We further note that often only a single covariate and dependent variable is available. Thus, some model for the efficient analysis of such data sets is needed.

The weighted ANCOVA methodology was developed from Neter et al. (1985), Snedecor and Cochran (1980), Press (1972), Armitage (1971), Bliss (1970), and Li (1965). In an ANCOVA for contaminant concentrations, year-means of Y are adjusted to a common value, A, of the covariate, X. These adjusted means are then analyzed (a) to test the general null hypothesis (H_{0A}) that these means are all equal, and (b) to test the specific null hypothesis H_{0L} of no time trend. Any specific value, A, may be chosen for the covariate, X. The most frequently chosen value is the grand mean of the covariate (Li 1965). An ideal example is shown in Figure 1. The adjusted mean of Y for group (year) j is simply the ordinate of its regression line at $X = A$. In the general case where K regression lines are not parallel, e.g. Figure 2, (i.e., variation from group to group in b_j is not within the sampling error), the year effect can only be studied by comparing the regression lines for each year (Neter et al. 1985) and the test of H_{0A} that adjusted year-means are all equal would depend upon the value of A selected.

Two weighting procedures are reported (Misra et al. 1990), based on results of tests of equalities of slopes and variances, in particular, a general one for use when heterogeneity of σ_j^2 and β_j values exists. The test for a time trend is as follows:

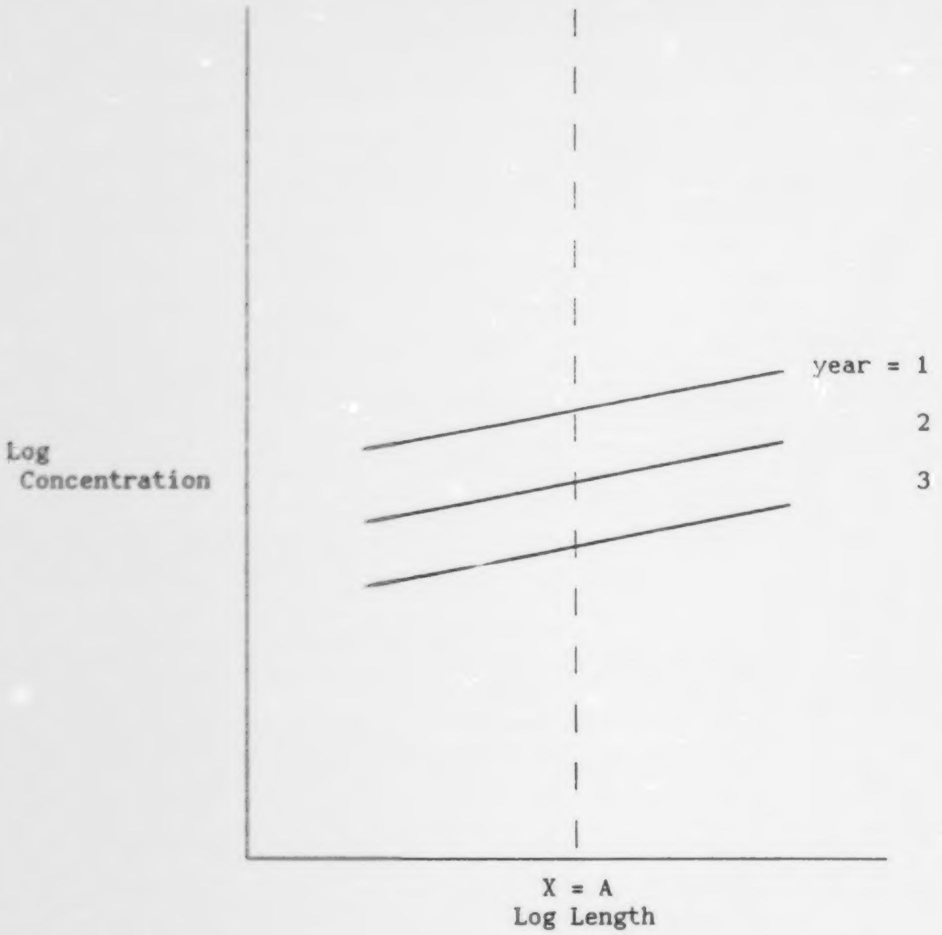

FIGURE 1. Regression lines for K groups (years) – Ideal Situation.

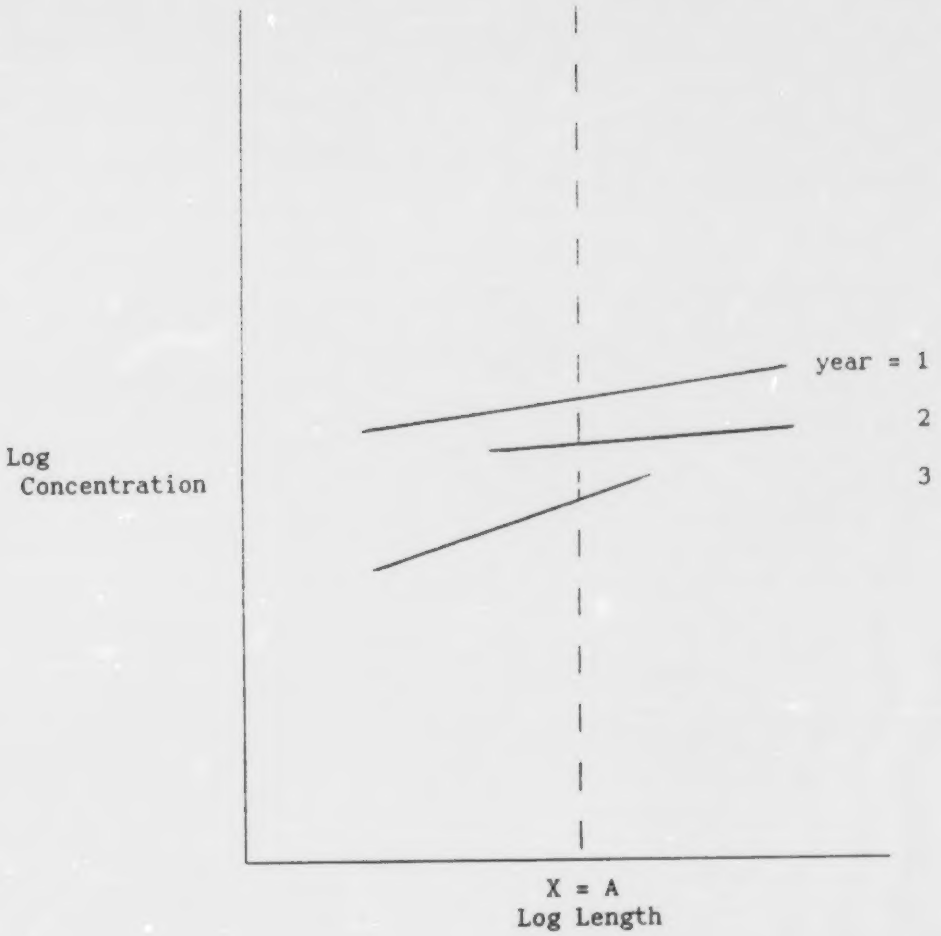

FIGURE 2. Regression lines for K groups (years) – Example of Real Situation

Define linear comparison

$$\hat{Z} = \sum_{j=1}^{K} M_j \bar{Y}_{Aj} \qquad \text{where} \quad \sum_{j=1}^{K} M_j = 0$$

for a time trend by providing appropriate values of M_j from a table of polynomials where years are equally spaced, or by calculating their values by the method explained in Bliss (1970) when years are not equally spaced. Calculate SS due to this linear comparison as

$$SSL = \left(\sum_{j=1}^{K} w_{Aj} M_j \bar{Y}_{Aj} - \sum_{j=1}^{K} w_{Aj} M_j \cdot \sum_{j=1}^{K} w_{Aj} \bar{Y}_{Aj} / \sum_{j=1}^{K} w_{Aj}\right)^2 / \left(\sum_{j=1}^{K} w_{Aj} M_j^2 - \left(\sum_{j=1}^{K} w_{Aj} M_j\right)^2 / \sum_{j=1}^{K} w_{Aj}\right)$$

Null hypothesis H_{0L}: $Z = 0$ for time trend is then tested.

The procedure was applied to a set of time trend data (flounder and cod from the Belgian coast). As heterogeneity of σ_j^2 and β_j values existed, temporal variations of adjusted year-means of Y and their time trends were analyzed by the general weighted method. Figure 3 gives time trendlines for the six data sets. The following benefits were found:

1. All S.E. values of the common regression coefficients were smaller when weights (equation 9) were employed.

2. All common regression coefficients except the regression coefficient for Y_2 of cod were significant when weights were employed.

MULTIPLE ANALYSIS OF COVARIANCE USING PC-MLR APPROACH

Neter et al. (1985) and Draper and Smith (1981) discuss the problem of correlated regressor (independent) variables. Although all possible regressor variables are desirable in the MLR, the use of a large set of variables presents a multitude of difficulties, including the occurrence of highly significant, yet erroneous findings. The use of principal components (PCs) as working variables offers several advantages. There is no loss of information if the set of X_j variables is replaced by their PCs (Harris 1975). Frequently, with biological data the information underlying the p variables X_j is summarized satisfactorily in a small number (q) of their PCs. PC analysis is provides a means of discovering the fact of linear dependence of X_j variables and specifying the nature of the dependence. In PC-MLR with q PCs the combined effect of q partial regression coefficients is partitioned orthogonally. Thus PCs can be used as covariates either singularly or simultaneously in the MANCOVA.

The MANCOVA is explained in Srivastava and Carter (1983) and Morrison (1976) and employed to analyze data on Canadian cod in Misra and Uthe (1987). The PC-MLR approach is described in Misra et al. (1989) with a full account given in Johnson and Wichern (1988), Draper and Smith (1981), and Morrison (1976). PCs were developed from the covariance matrix S of X_j with p eigen value-eigen vector (L_K - Z_K) pairs. The principal component transformation is an orthogonal transformation which rotates the coordinate axis of the original X_j variables to the axes of the PCs which have the following features:

1. These PCs are uncorrelated with each other;

2. Variability of the total system is associated with PCs in decreasing order.

Therefore, it is possible that the first one or two PC variables are sufficient to summarize the bulk of the variability and covariability of the original X_j variables. PC coefficients are scaled to make Z_K of unit length. Draper and Smith (1981) and Morrison (1976) suggest that the first few Z_Ks which explain 75% or more of the total variance will adequately replace the p variables X_j.

The approach was applied to Canadian cod contaminant concentration and burden (concentration times the weight of the whole tissue) data (Misra et al. 1989). For concentrations data, two eigen values explained 81.4% of the total (standardized sample) variance, while 82.6% of the total standardized variance in the burden data was explained by the first eigen value. Linear time trends were significant although significant deviations from the linear trend were also present.

DISCUSSION

Trend analysis by either of the two above methods was shown to be superior to the more common methods of ANCOVA or MLR with multiple independent variables. The PC-MLR approach offers a method of overcoming

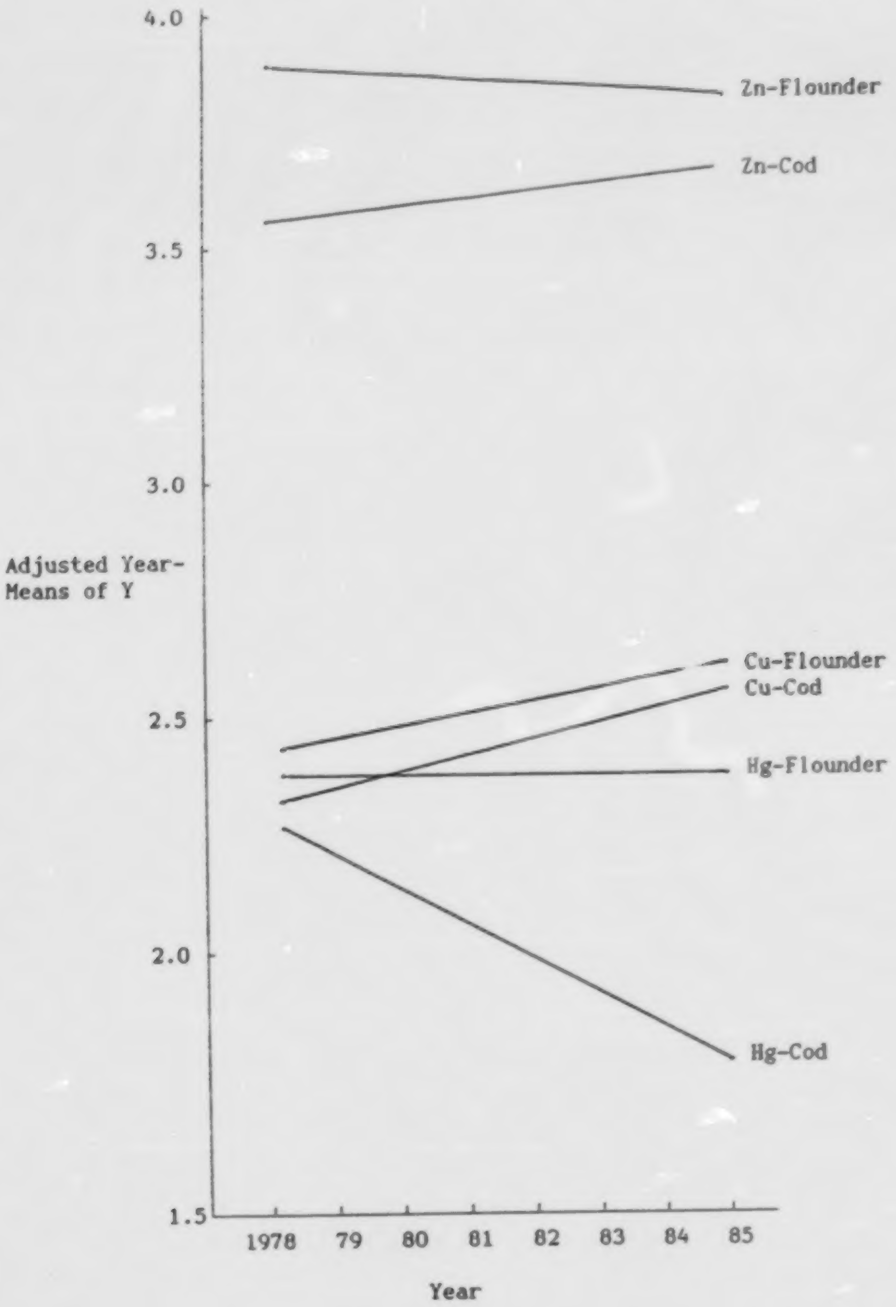

FIGURE 3. Time trends for Zn, Cu and Hg in Cod and Flounder of the Belgian coast, 1978–1985.

problems associated with multicollinearity as well. Analysis of real data sets showed the feasibility of detecting significant time trends of relatively low magnitude within a population of rather diverse characteristics.

REFERENCES

Armitage, P. 1971. Statistical Methods in Medical Research. Blackwell Scientific Publications, Oxford, UK.

Bliss, C.I. 1970. Statistics in Biology. Vol. 2. McGraw-Hill, New York, NY.

Draper, N.R., and H. Smith. 1981. Applied Regression Analysis. John Wiley and Sons, Inc. New York, NY.

Falconer, D.S. 1972. Introduction to Quantitative Genetics. Oliver and Boyd, Edinburgh.

Harris, R. A. 1975. A Primer of Multivariate Statistics. Academic Press, New York, NY.

International Council for the Exploration of the Sea. 1986. Report of the meeting of the *ad hoc* Group of Statisticians Assisting the Working Group on Marine Pollution Baseline and Monitoring Studies in the North Atlantic on Trend Monitoring. C.M. 1986/E:39.

International Council for the Exploration of the Sea. 1987a. Report of the ICES Advisory Committee on Marine Pollution, 1986. ICES Coop. Res. Rep. No. 142: 128 p.

International Council for the Exploration of the Sea. 1987b. Report of the 1987 meeting of the Working Group on Statistical Aspects of Trend Monitoring. C.M. 1987/-E:24.

International Council for the Exploration of the Sea. 1989. Statistical analysis of the ICES cooperative monitoring programme data on contaminants in fish muscle tissue (1978-1985) for determination of temporal trends. Coop. Res. Rep. 162, 147 p.

Johnson, J.R., and D.W. Wichern. 1988. Applied Multivariate Statistical Analysis. Prentice-Hall, Englewood Cliffs, NJ.

Li, J. C. R. 1965. Statistical Inference: Vol. 1. Edwards Brothers, Ann Arbor, MI.

Miller, R.G. 1981. Simultaneous Statistical Inference. Springer-Verlag, New York, NY.

Misra, R.K. and J.F. Uthe. 1987. Methods of time trends analysis applied to contaminant levels in Canadian Atlantic cod (*Gadus morhua*). Can. J. Fish. Aquat. Sci. 44: 859-865.

Misra, R.K., J.F. Uthe, D.P. Scott, C.L. Chou, and C.J. Musial. 1989. Time trends of chemical contaminant levels in Canadian Atlantic cod (*Gadus morhua*) with several biological variables. Mar. Poll. Bull. 20(5): 227-232.

Misra, R.K., J.F. Uthe, and W. Vyncke. 1990. A weighted procedure of analyzing for time trends in contaminant levels in CMP data: Application to cod and flounder data of the Belgian coast (1978-1985). J. Cons. int. Explor. Mer. 47: 65-75.

Morrison, D.F. 1976. Multivariate Statistical Methods. McGraw-Hill Book Company, New York, NY.

Neter, J., W. Wasserman, and M.H. Kutner. 1985. Applied Linear Statistical Methods. Richard D. Irwin, Homewood, NJ.

Nicholson, M.D., and S.J. Wilson. 1987. Identification of trends in levels of metals in fish muscle: Appraisal of the statistical analysis and of data quality. ICES C.M. 1987/E:24, Annex 3.2, p. 16-21.

Press, S.J. 1972. Applied Multivariate Analysis. Holt, Rinehart and Winston Inc., New York NY.

Snedecor, G.W., and Cochran, W.G. 1980. Statistical Methods. Iowa State University Press, Ames, IA.

Srivastava, M.S., and E.M. Carter. 1983. An Introduction to Applied Multivariate Statistics. North-Holland, New York, NY.

MARINE MAMMALS AS INDICATORS OF ENVIRONMENTAL CONTAMINATION BY PCBs AND DIOXINS/FURANS

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INTRODUCTION

It has been recognized since the late 1960's that marine mammals can accumulate high concentrations of lipophilic organochlorine pollutants in blubber and that monitoring of these populations would assist in identifying areas of contamination (Holden 1972). In Canada, detailed studies of PCB and organochlorine (OC) pesticides residues in marine mammal tissues began in the early 1970's with investigations of concentrations in Gulf of St. Lawrence harp seals (Addison et al. 1973; Jones et al. 1976) and Bay of Fundy harbour porpoise (Gaskin et al. 1971). Ringed seals and beluga in the western Canadian Arctic were also investigated (Addison and Smith 1974; Addison and Brodie 1973). Compared to ringed seals in the Baltic Sea (Helle et al. 1976) and harbour seals from the Dutch Wadden Sea (Reijnders, 1980), harp and ringed seals had low levels of PCBs and DDT-related compounds. Harbour porpoise from the Bay of Fundy, had among the highest DDT levels ever reported in cetaceans, probably resulting from the use of DDT for forest spraying during the 1960's in New Brunswick and Maine.

These early studies dealt mainly with DDT and PCBs and was limited by the technology available (packed column gas chromatography (GC)) to separate individual isomers of PCBs and OC pesticides. Over the past five years our laboratories have collaborated in the analysis of marine mammal samples from Canadian waters, using high resolution capillary GC. The list of OC contaminants found in marine mammal tissues now includes about 60 PCB congeners, as well as chlordane- and DDT-related compounds, toxaphene or polychlorinated camphenes (PCCs), mirex, and chlorobenzenes, hexachlorocyclohexanes, and polychlorinated dibenzo-g-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs).

RESULTS

Results for PCBs, two major OC pesticides, and three major PCDD/PCDF congeners are summarized in Table 1. The summary includes most of Canada's resident cetacean (odontocetes) populations as well as Arctic ringed seals and polar bears.

Belugas from the St. Lawrence River estuary have the dubious distinction of having the highest PCB levels of all Canadian marine mammals analysed to date. DDT and PCCs were also prominent OCs in beluga fat, however, PCDDs and PCDFs are near detection limits (Table 1). Killer whales from Georgia Strait/Vancouver Island have the second highest PCB levels and are distinguished from beluga by higher levels of 2,3,7,8-TCDF. A false killer whale from Georgia Strait had 1520 µg/9 total DDT (ΣDDT), mainly as the metabolite 4,4'-DDE, the highest level in any of the marine mammals analysed.

Harbour porpoise from Georgia Strait also had the highest PCDD/PCDFs (Table 1). The highest level of 2,3,7,8-TCDD in the marine mammals surveyed was found in blubber of ringed seals from Barrow Strait in the central Arctic archipelago. White-beaked dolphins from the Gulf of St. Lawrence also had high DDT, PCC and PCB levels.

DISCUSSION

The results are discussed below in relation to their usefulness in assessing geographic and temporal trends in contaminants, and biological effects, and the capability of marine mammals to biotransform some contaminants.

Information on levels of contaminants in marine mammals tissues is perhaps most useful as an indicator of general geographic and temporal trends of contaminants in marine food chains. The results for polar bears in Table 1 are from a larger study of PCBs and OC pesticides in tissues of polar bears from 12 management zones in the Canadian Arctic (Norstrom et al. 1988). Polar bears feed almost exclusively on seals, and their distribution is well known, so they are an excellent species for characterizing marine contamination. Levels of most organochlorines were lowest in the high Arctic and highest in Hudson Bay polar bears reflecting the relative isolation of the high Arctic groups from North American sources of contaminants (Table 1).

Addison et al. (1984;1986) have used east coast grey seals and Arctic ringed seals to study temporal trends in PCBs and DDT-group compounds. Care was taken to match the samples in terms of age, sex, and blubber thickness. They found significant declines in total DDT (ΣDDT) but not for PCBs in Grey seals over the period 1974 to 1982. Levels of PCBs declined in Arctic ringed seals over the period 1972

Table 1. Organochlorine contaminants in marine mammal fat (fresh weight) from Canadian waters.

Species	N	Sex[1]	Location	Mean concentration ($\mu g\ g^{-1} \pm SD$)			Mean Concentration (ng/kg)			References[4]
				ΣDDT	PCC	ΣPCB[2]	TCDD	TCDF	HxCDD[3]	
Polar bear	18	P	Cornwallis Is.	0.30	ND[5]	5.94	20[6]	<2	<4	A,B
	10	P	North Baffin Is.	0.21	ND	4.22	4	<2	<4	
	10	P	W. Davis Strait	0.39	ND	3.24	3	<2	<4	
	20	P	South Baffin Is.	0.41	ND	4.25	5	<2	<4	
	9	P	W. Hudson Bay	1.19	ND	8.02	2	<2	<4	
Ringed seal	10	M	Cumberland Sound	0.33 ± 0.19	0.38 ± 0.16	0.51 ± 0.23	8	4	<8	A,C
	8	M	W. Davis Strait	0.41 ± 0.19	0.25 ± 0.16	0.54 ± 0.21	11	3	<8	
	16	M	Barrow Strait	0.71 ± 0.41	ND	0.57 ± 0.29	37	4	9	
Narwhal	16	M	Pond Inlet (Baffin Bay)	5.92 ± 1.71	12.2 ± 2.44	5.18 ± 1.34	ND	ND	ND	D
Beluga	6	M	Cumberland Sound	6.83 ± 1.89	13.1 ± 3.75	4.91 ± 0.25	<2	<2	5	A,C
	8	M	Jones Sound (Baffin Bay)	1.96 ± 0.32	4.25 ± 1.02	2.53 ± 0.57	ND	ND	ND	
	11	M	St. Lawrence estuary	96.1 ± 56.1	25.5 ± 4.14	85.5 ± 63.3	<2[7]	2	2	E,F
White-beaked dolphin	9	M	Gulf of St. Lawrence	43.4 ± 26.7	46.0 ± 22.1	34.2 ± 22.4	ND	ND	ND	G
Pilot whale	5	M	Newfoundland S. Coast	11.9 ± 6.10	11.7 ± 7.05	9.03 ± 3.80	ND	ND	ND	G
Killer whale	6	M+F	Georgia Str./Van. Is.	50.4 ± 52.3	12.5 ± 14.1	31.4 ± 29.3	<2[8]	19	6	F,H
Harbour porpoise	7	M+F	Georgia Str./Van. Is.	13.0 ± 8.4	5.7 ± 3.0	11.3 ± 6.9	2	20	46	F,H
False killer whale	2	M	Georgia Str./Van. Is.	75.9, 1920	16.6, 89.2	45.2, 33.9	<2, 8	2, 109	2, 23	F,H

[1] P = pooled samples, M= males, F = females.

[2] Total PCB congeners except as indicated.

[3] 1,2,3,6,7,8-hexachlorodibenzo-p-dioxin.

[4] References: A, Norstrom et al. (1990); B, Norstrom et al. (1988);C, Muir et al. (1988a);D, Muir et al. (1991); E, Muir et al. (1990); F, Norstrom and Simon (1990); G, Muir et al. (1988b); H, Muir. Unpublished results 1990.

[5] Not determined.

[6] Results PCDDs and PCDFs for liver, all other for fat samples. Pooled samples were analysed except for St. Lawrence beluga and West coast killer whales.

[7] PCDD/PCDF results based on mean of 10 oil samples (M + F) assuming undetectable levels are one-half of the detection limit.

[8] PCDD/PCDF results are means from 6 killer whale blubber samples with similar assumptions as footnote 7.

to 1981 while declines of DDT-related compounds were not statistically significant.

There are obvious limitations to the use of marine mammals for determining geographic and temporal trends. Sex and age are important sources of variation (Aguilar 1987). Females are generally less contaminated than males because of excretion of lipophilic organochlorines via lactation. Age of sexual maturity and duration of active parturition has been determined in Dall's porpoise based on the relationships of PCBs and 4,4'-DDE to age (Subramanian et al. 1988). Recalcitrant organochlorines such as hexa- to nonachloro- PCBs, 4,4'-DDE and *trans*-nonachlor (a component of chlordane) are usually significantly correlated with age in male seals (Addison and Smith 1973; Muir et al. 1988) and toothed and baleen whales (Tanabe et al. 1987; Aguilar and Borrell, 1988).

Age and sex can be taken into account in geographic and temporal comparisons by selecting age classes of the same sex, but factors such as lack of knowledge of diet of the animals, availability of stranded versus hunted animals, and differences in analytical methodology (including technological changes in detection over time) may also influence the conclusions of such studies. In the Canadian Arctic adult male narwhals and belugas from the Davis Strait and Baffin Bay region had significantly higher ΣPCBs, ΣDDT and PCCs than animals of approximately the same age in Hudson Bay and the Beaufort Sea (Table 1). The reasons for this are not known but dietary differences may be important because narwhal are known to prey on Greenland halibut, a deep sea predator, found in Davis Strait but not present in the shallow waters of Hudson Strait or the Beaufort Sea (Muir et al. 1991). St. Lawrence beluga can be distinguished from other beluga stocks and from cetaceans in the Gulf of St. Lawrence by higher levels of mirex (Muir et al. 1990). Although the diet of the beluga is varied, including benthic organisms as well as pelagic invertebrates and fish, levels of mirex in the animals can be accounted for by assuming that eels, which migrate annually from Lake Ontario to the Sargasso Sea, form a small.portion (<10%) of the diet (Beland and Mirtineau 1988).

Although samples from stranded animals are commonly used in studies of contaminants in marine mammals there is always the question of how representative they are of the population. Levels of organochlorines in tissues may vary because of loss of lipid and transformation of the contaminants in the decomposing tissues (Borrell and Aguilar 1990). Bergman et al. (1981) found that PCB levels in dead, stranded ringed and Grey seals from the Baltic Sea were higher than in live (hunted) animals and that levels did not correlate with age. Results from the St. Lawrence beluga and west coast killer whales (Table 1) are all from stranded animals and may be subject to these problems. However, St. Lawrence beluga have several characteristics in common with their (hunted) Arctic relatives such as similar blubber lipid levels, lower contaminant levels in females than in males and positive correlations of tissue concentrations in males with age. On the other hand, stranded female beluga from the St. Lawrence are older than those from the Arctic and have positive correlations of PCBs and ΣDDT with age, indicating a lower frequency of parturitions (Muir et al. 1990).

A further complication in interpreting survey of OCs levels in marine mammals, especially between species, is metabolism of the contaminants. The pattern of PCB congeners in odontocetes is similar; penta- and hexachlorobiphenyls predominate and trichlorobiphenyls are virtually absent. This contrasts with most marine fish and invertebrates in which tri- and tetrachlorobiphenyls usually predominate (Duinker and Hillebrand 1983; Muir et al. 1988a). The 2,5- and 2,3,6-substituted tetra-, penta- and hexachloro-congeners are more prominent in cetaceans and pinnipeds than in terrestrial mammals (Tanabe et al. 1988). Polar bears have the most remarkable capability to metabolize PCBs, DDT and chlordanerelated compounds. Three congeners, all with 2,4,5- and 2,3,4,5-substitution, accounted for 71% of ΣPCB in polar bear fat (Norstrom et al. 1988).

Marine mammals appear to have a wide range of capabilities to metabolize PCDDs and PCDFs. Arctic ringed seal feeding in the same region as beluga and polar bear had higher levels of 2,3,7,8-TCDD, TCDF and OCDD than the latter species, although they had from 5 to 10-fold lower levels of PCBs (Table 1). Buckland et al. (1990) detected only 2,3,7,8-substituted PCDDs and PCDF congeners in Hector's dolphin from New Zealand, the usual finding for birds, fish and terrestrial mammals. The St. Lawrence beluga had undetectable 2,3,7,8-TCDD but did contain low ng/kg levels of 1,2,4,7,8-PnCDF, 1,2,4,6,7,8-, 1,2,4,6,8,9--HxCDF. Dall's porpoise from Georgia Strait/Vancouver Island also contained the higher chlorinated non-2,3,7,8-substituted PCDFs while killer whales, from the same waters, did not (Norstrom and Simon, unpublished data 1990). The most prominent PCDD in marine mammals from Georgia Strait was 1,2,3,6,7,8-HxCDD (Table 1). The source of this contaminant is probably the use chlorophenols and of chlorophenol contaminated wood chips in bleached kraft mills.

Studies of the patterns of PCBs and PCDD/PCDF congeners give some insight into marine mammal physiology. Whales have relatively low levels of metabolic activity towards phenobarbital-type substrates, relative to rats, but high levels of activity of isozymes that metabolize PAHs (Watanabe et al. 1989). The consequences of elevated mixed function oxidase activity at the level of individual animals and populations are unknown, however, altered physiological processes, i.e. reproductive failures, have been associated with high levels of PCBs in some marine mammal populations (Helle et al. 1976; Reijnders 1980; Olsson 1986; Martineau et al. 1987).

As top predators with long life spans, marine mammals should be good indicators of eco system "health". However direct cause and effect relationships between levels of contaminants and effects such as reproductive failure have been difficult to establish (Reijnders 1986a; Addison 1989). This is a consequence of the lack of information on biochemical and physiological effects of the contaminants in marine mammals, and because of the mixture of pollutants to which the animals are exposed to in their diet. Studies with mink and with captive habour seals have shown reproductive failure can be induced on a fish diet high in PCBs (Reijnders 1986b; Bleavins et al. 1980). Recent studies have identified non-ortho-substituted PCBs as the toxicants inducing reproductive failures in mink (Olsson et al. 1990).

In conclusion, marine mammals are good indicators of geographic and temporal trends of recalcitrant OCs such as highly chlorinated PCB congeners, chlordane and DDT-related compounds, provided that effects of age, sex and diet are considered. Cetaceans appear to have a wide range of capability to metabolize PCDD/PCDFs and are therefore not good indicators for geographic trends of these components. Marine mammals can also serve as indicators of ecosystem health but direct cause and effect relationships between observed biological effects and contaminant levels remain to be established.

ACKNOWLEDGMENTS

We are grateful to Dr. Ken Langelier (Nanaimo, B.C.) for providing tissues of killer whales and harbour porpoises, to Dr. Pierre Béland (St. Lawrence National Institute of Ecotoxicology, Rimouski, QU) for beluga samples and to Dr. J. Lien (Memorial University, St. John's, NF) for dolphin and pilot whale samples. PCB and OC pesticides analyses were performed by Bruno Rosenberg, Cathy Ford and Bert Grift, and PCDD/PCDF analyses by Mary Simon, Mike Mulvihill and Mary Kleiber.

REFERENCES

Addison, R.F. 1989. Organochlorines and marine mammal reproduction. Can. J. Fish. Aquat. Sci. 46: 360-368.

Addison, R.F. and P.F. Brodie, 1973. Occurrence of DDT residues in beluga whales (*Delphinapterus leucas*) from the Mackenzie Delta, N.W.T. J. Fish. Res. Board Can., 30: 1733-1736.

Addison, R.F., S.R. Kerr, J. Dale, and D.E. Sergeant. 1973. Variation of organochlorine residue levels with age in Gulf of St. Lawrence harp seals (*Pagophilus groenlandicus*). J. Fish. Res. Board Can. 30: 595-600.

Addison, R.F. and T.G. Smith. 1974. Organochlorine residue levels in Arntic ringed seals: variation with age and sex. Oikos 25:335-337.

Addison, R.F., P.F. Brodie, M.E. Zinck and D.E. Sergeant. 1984. DDT has declined more than PCBs in eastern Canadian seals during the 1970's. Environ. Sci. Technol. 18: 935-937.

Addison, R.F., M.E. Zinck, and T.G. Smith. 1986. PCBs have declined more than DDT-group residues in arctic ringed seals (*Phoca hispida*) between 1972 and 1981. Environ. Sci. Technol. 20: 253-256.

Aguilar, A. and A. Borrell. 1988. Age- and sex-related changes in organochlorine compound levels in fin whales (*Balaenoptera physalus*) from the Eastern North Atlantic. Mar. Environ. Res. 25: 195-211.

Aguilar, A. 1987. Using organochlorine pollutants to discriminate marine mammal populations. A review and critique of methods. Mar. Mamm. Sci. 3: 242-263.

Béland,P. and Martineau, D. (1988). The beluga whale (*Delphinapterus leucas*) as integrator of the St. Lawrence basin contamination history. In: Proceedings of the Intern. Conference on Bio-indicators: Exposure and Effects. Oak Ridge National Laboratory, Knoxville, TN. March 20-23, 1988.

Bergman, A., M. Olsson and L. Reutergårdh. 1981. Lowered reproduction rate in seal population and PCB. A discussion of comparability of results and a presentation of some data from research on the Baltic seals. Int'l Coun. Explor. Sea., Mar. Mamm. Comm. C.M. 1981/N:10. 18 pp.

Bleavins, M.R., R.J. Aulerich, and R.K. Ringer. 1980. Polychlorinated biphenyls (Aroclors 1016 and 1242): Effects on survival and reproduction in mink and ferrets. Arch. Environ. Contam. Toxicol. 9: 627-635.

Borrell, A. and A. Aguilar. 1990. Loss of organochlorine compounds in the tissues of a decomposing stranded dolphin. Bull. Environ. Contam. Toxicol. 45: 46-53.

Buckland, S.J., D.J. Hannah, J.A. Taucher, E. Slooten and S. Dawson. 1990. Polychlorinated dibenzo-p-dioxins and dibenzofurans in New Zealand's Hector's dolphin. Chemosphere 20: 1035-1042.

Duinker, J.C. and M.T.J. Hillebrand, 1983. Composition of PCB mixtures in biotic and abiotic marine compartments (Dutch Wadden Sea). Bull. Environ. Contam. Toxicol. 31:25-32.

Gaskin, D.E., M. Holdrinet, and R. Frank. 1972. 1971. Organochlorine pesticide residues in harbour porpoises from the Bay of Fundy region. Nature 233:499-500.

Helle, E., M. Olsson and S. Jensen. 1976. DDT and PCB levels and reproduction in ringed seal from the Bothnian Bay. Ambio 5:188-189.

Holden, A.V. 1972. Monitoring organochlorine contamination of the marine environment by the analysis of residues in seals. In: Marine Pollution and Sea Life, M. Ruivo, Ed., Fishing News Books Ltd., England. pp. 266272.

Jones, D., K. Ronald, D.M. Lavigne, R. Frank, M. Holdinret and J.F. Uthe. 1976. Organochlorine and mercury residues in the harp seal (*Pagophilus groenlandicus*). Sci. Total Environ. 5: 181-195.

Martineau, D., P. Béland, C. Desjardins, and A. Lagacé. 1987. Levels of organochlorine chemicals in tissues of beluga whales (*Delphinapterus leucas*) from the St. Lawrence estuary, Québec, Canada. Arch. Environ. Contam. Toxicol. 16: 137-147.

Muir, D.C.G., R. Wagemann, N.P. Grift, R.J. Norstrom, M. Simon, and J. Lien. 1988a. Organochlorine chemical and heavy metal contaminants in white-beaked dolphins (*Lagenorhynchus albirostris*) and pilot whales (*Globicephala melaena*) from the coast of Newfoundland, Canada. Arch. Environ. Toxicol. Chem. 17: 613-629.

Muir, D.C.G., R.J. Norstrom, and M. Simon. 1988b. Organochlorine contaminants in arctic marine food chains: Accumulation of specific polychlorinated biphenyls and chlordane-related compounds. Environ. Sci. Technol. 22: 1071-1079.

Muir, D.C.G., C.A. Ford, R.E.A. Stewart, T.G. Smith, R.F. Addison, M.E. Zinck and P. Béland. 1990. Organochlorine contaminants in belugas (*Delphinapterus leucas*) from Canadian waters. In: Advances in research on the beluga whale Delphinapterus leucas. T.G. Smith, D.J. St. Aubin and J.R. Geraci, Eds., Can. J. Fish. Aquat. Sci. Bull. No: xx In press.

Muir, D.C.G., C.A. Ford, N.P. Grift, R.E.A. Stewart, and T.F. Bidleman, 1991. Organochlorine contaminants in narwhal (*Monodon monoceros*) from the Canadian arctic. Submitted to Environ. Pollut.

Norstrom, R.J., M. Simon, D.C.G. Muir, and R.E. Schweinsburg. 1988. Organochlorine contaminants in arctic marine food chains: Identification, geographical distribution and temporal trends in polar bears. Environ. Sci. Technol. 22: 1063-1071.

Norstrom, R.J. and M. Simon. 1990. Levels and possible sources of polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) in beluga whale tissues collected from the St. Lawrence River in 1987 and 1988. Report CRD 90-9. Canadian Wildlife Service, Environment Canada, Ottawa. 6 pp.

Norstrom, R.J., M. Simon and D.C.G. Muir. 1990. Chlorinated dioxins and furans in marine mammals from the Canadian arctic. Environ. Pollut. 66, 1-20.

Olsson, M. 1986. PCBs in the Baltic environment. In: PCBs in the Environment, J.S. Waid (Ed), Vol. III, CRC Press, Boca Raton, Fl pp. 181-298.

Olsson, M., A. Bergman, S. Jensen and J.E. Kihlström. 1990. Effects of various fraction of PCB on mink reproduction. Preliminary results from experimental studies within the Swedish seal project. Proceedings of Dioxin 1990. Vol. 1. pp. 393-396.

Reijnders, P.J.H. 1980. Organochlorine and heavy metal residues in habour seals from the Wadden Sea and their possible effects on reproduction. Neth. J. Sea Res. 14: 130-165.

Reijnders, P.J.H. 1986a. Reproductive failure in common seals feeding on fish from polluted coastal waters. Nature 324, 456-457.

Reijnders, P.J.H. 1986b. Perspectives for studies of pollution in cetaceans. Mar. Pollut. Bull. 17: 58-59.

Subramanian, AN., S. Tanabe, S. and R. Tatsukawa. 1988. Use of organochlorines as chemical tracers in determining some reproductive parameters in Dalli-type Dall's porpoise (*Phocoenoides dalli*). Mar. Environ. Res. 25: 161-174.

Tanabe, S., B.G. Loganathan, An. Subramanian and R. Tatsukawa. 1987. Organochlorine residues in short-finned pilot whales. Possible use as tracers of biological parameters. Mar. Pollut. Bull. 18: 561-563.

Tanabe, S., S. Watanabe, H. Kan, R. Tatsukawa. 1988. Capacity and mode of PCB metabolism in small cetaceans. Mar. Mamm. Sci. 4: 103-124.

Watanabe, H., T. Shimada, S. Nakamura, N. Nishiyama, N. Yamashita, S. Tanabe and R.Tatsukawa. 1989. Specific profile of liver microsomal cytochrom P-450 in dolphin and whales. Mar. Environ. Res. 27: 51-65.

LONG-TERM EXPOSURE OF *NEANTHES* TO TOXIC MARINE SEDIMENT

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ABSTRACT

The effect of long-term exposure of juvenile *neanthes arenaceodentata* to toxic marine sediments was examined to determine the relationship between changes in juvenile biomass and endpoints associated with reproductive success. At the end of a 25 day exposure period, the highest average individual biomass reported was for the West Beach (WB) control and Carr Inlet (CI) reference sediments, with the lowest biomass reported for Elliot Bay (EB). As the amount of contamination (determined by the EB fraction in the test sediment) increased, biomass decreased to almost zero growth in the most contaminated sediment (100 percent). Delay in reaching sexual maturity increased as the contamination increased. Worms exposed to WB and CI sediments matured between days 20 and 25, whereas worms exposed to the EB or an dual volume mixture of CI and EB sediments (CI/EB) matured between days 25 and 32. EB male and female worms exhibited significant mortality during the adult stage. During the 63 day period over which the experiment was conducted, one WB pair and 50 percent of the CI pairs produced egg cases. None of the CI/EB or EB pairs reproduced. The results of this study indicate that the level of contamination affecting juvenile growth in is similar to the level that affects reproductive success.

INTRODUCTION

A sublethal sediment bioassay, using the juvenile stage of *Neanthes* has been developed and tested with Puget Sound sediments. Toxicity was determined by measuring differences in worm biomass and survival relative to a control following a 20-day exposure period. The sublethal bioassay is primarily designed to determine the effect of exposure to contaminated sediment on the growth of the juvenile life history stages of *Neanthes*. One approach to assessing the utility of growth as a sublethal measure of organism health is to evaluate the relationship in reduction in growth of the juvenile stage and eventual changes in other ecologically important endpoints such as reproduction. Changes in growth rate (e.g., ability to reach a specific life stage at a given time and within a range of size) can play a significant role in determining the reproductive success of individual organisms and populations. This study was conducted to determine the relationship between juvenile growth in *Neanthes* and reproductive success following exposure to contaminated sediments.

METHODS

The long-term experiment was conducted with sediments collected from Elliot Bay (EB), and Carr Inlet (CI) in Puget Sound, Washington State, U.S.A. The control sediment was obtained from West Beach (WB), a relatively uncontaminated site on Widbey Island, Washington State. In addition to the field collected sediments, a 50/50 mixture by wet weight) of EB and CI sediments (CI/EB) was included in the experiment to provide a gradient in sediment chemical contamination between the reference (CI) and contaminated (EB) sediments.

The long-term exposure experiment was divided into two phases, which were initiated simultaneously. The first phase of the experiment determined the growth response of *Neanthes* juveniles exposed to a 2 cm layer of test sediment for 10, 15, 20, and 25 days. The protocol described by Johns et al. (1990) was followed except that three replicates from each sediment treatment were taken at the four sampling periods to determine juvenile biomass and survival. The growth response following each exposure period was determined by measuring the increase in biomass of individual worms dried to a constant weight.

The second phase of the experiment determined the effects of sediment exposure on reproductive success. Juvenile *Neanthes* were exposed to the sediment treatments until reaching sexual maturity employing the test protocol described by Johns et al. (1990). At sexual maturity, surviving worms from each treatment replicate were collected and ten worm pairs were selected for each treatment. Each worm pair was placed in a 3 – 4 mm layer of test sediment for the remainder of the experiment. Following 63 days of exposure the experiment was terminated. Response criteria used to evaluate reproductive success were adult worm survival, adult worm size (as dry weight), time to sexual maturity, time to deposition of an egg case, and the relative number of eggs in surviving females.

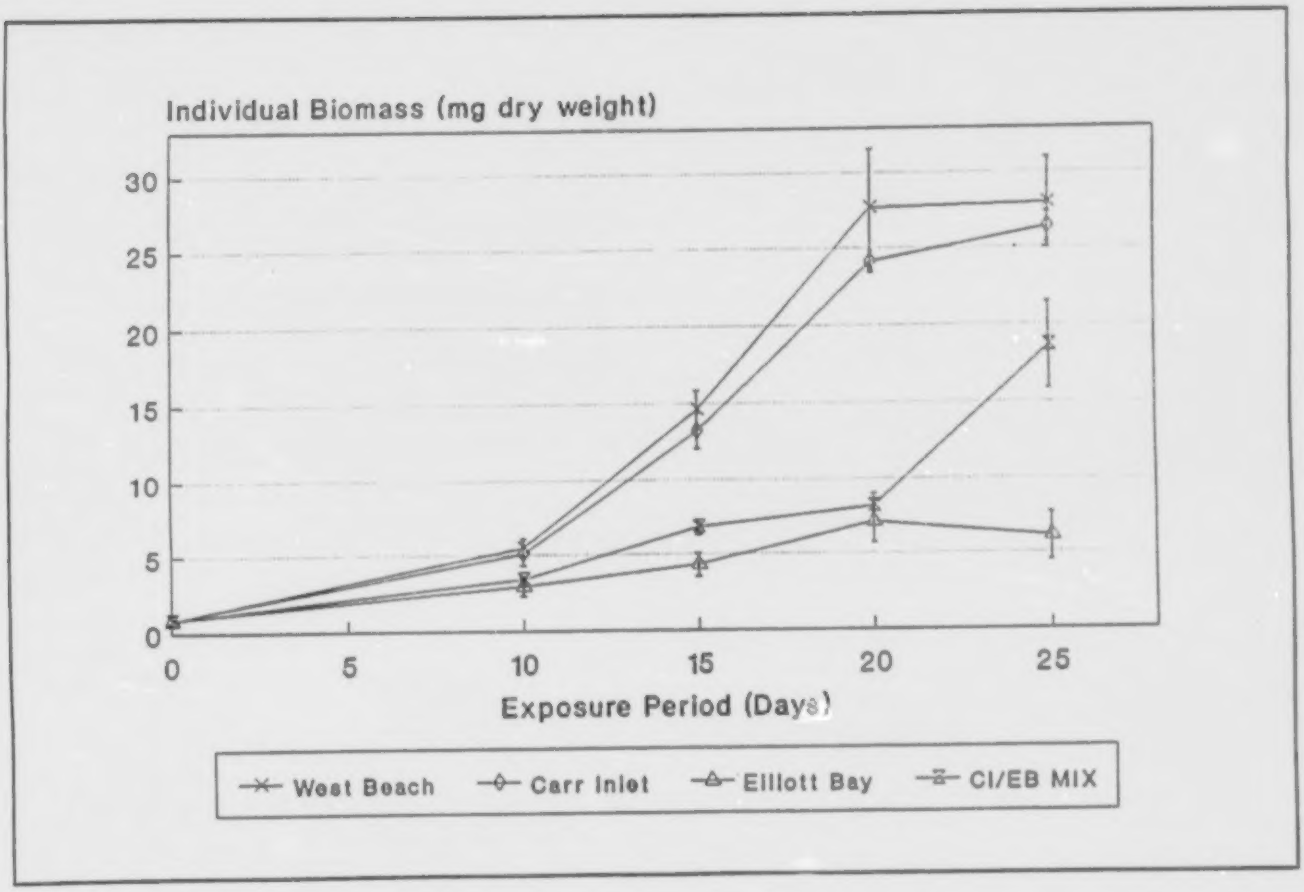

FIGURE 1. Growth rates and biomass of juvenile worms exposed to Puget Sound sediments. Data are presented as mean ± standard error.

Sediment Type	Time to Sexual Maturity (days)	Time to Egg Laying (days)	Percent Females Laying Eggs[a]	Percent Male Survival[a]	Percent Female Survival[a]	Dry Weight Surviving Males[a]	Dry Weight Surviving Females[a]	Proportion of Body with Eggs
West Beach	25	52	10	100	100	37.9 ± 2.5	92.5 ± 10.1	0.94
Carr Inlet	25	58	50	100	100	37.0 ± 2.8	93.4 ± 8.9	1.00
Carr Inlet/ Elliot Bay	32	--	0	100	100	27.9 ± 1.7	80.3 ± 7.6	0.95
Elliot Bay	32	--	0	40	50	11.7 ± 2.9	25.8 ± 3.8	0.58

[a] Following 63-day exposure.

TABLE 1. Characteristics of reporductive success in *Neanthes* exposed to different sediments.

RESULTS AND DISCUSSION

In the first phase of the long-term experiment, survival was greater than 93 percent in all four sediment treatments. Individual biomass increased with increasing exposure in all treatments with the greatest increase in biomass occurring between days 10 and 20 (Figure 1). A graded response in growth was observed at the end of the 25-day exposure period, with the highest average individual biomass reported for worms from the West Beach and Carr Inlet treatments, followed by worms from the CI/EB treatment. The lowest biomass was observed for worms from the EB treatment. The pattern of growth in worms from the CI/EB treatment was similar to that observed for worms from the EB treatment until day 20. Between day 20 and 25, however, a significant increase in biomass was observed in worms from the CI/EB treatment relative to worms from the EB treatment. Worms maintained in the EB treatment did not exhibit a significant increase in biomass during the 25 day exposure period. In summary, as the amount of contamination in the sediment (determined by the EB fraction) was increased, biomass decreased to the point of almost zero growth in the most contaminated sediment (EB).

A gradation in response was also observed in the response criteria used to assess reproductive success (Table 1). Worms exposed to WB and CI treatments developed to sexual maturity (between days 20 and 25) before worms exposed to the EB and CI/EB treatments (between days 26 and 32). Survival of the adult worm pairs was high (100 percent) in all treatments except the EB treatment. In the EB treatment, only 40 percent of the males and 50 percent of the females were surviving at the termination of the experiment (Table 1). In all four treatments the females were larger than males. Male worms from the WB, CI, and CI/EB treatments were significantly larger than males collected from the EB treatment (Table 1). Females maintained in the WB, CI, and CI/EB treatments were also significantly larger than females collected from the EB treatment (Table 1). At the termination of the experiment, 10 percent of the worm pairs from the WB treatment and 50 percent of the worm pairs from the CI treatment had reproduced. No egg mass was observed in any of the worm pairs from the CI/EB and EB treatments. Surviving females from the WB, CI and CI/EB treatments exhibited a greater relative egg density in the body cavity than did females maintained in the EB treatment. The relative proportion of the body cavity filled with eggs for the three treatments ranged from 94 to 100 percent, while only 58 percent of the body cavity of females from the EB treatment had eggs (Table 1). As with juvenile growth, reproductive success in *Neanthes* appears to be affected by an exposure to both the EB and CI/EB treatments.

The results of this study indicate that a level of sediment contamination causing significant effects on juvenile growth (ie., exposure to EB sediment) also affected reproductive success in adult *Neanthes*. However, at a lower level of contamination causing equivocal effects on juvenile worm growth (ie., exposure to CI/EB sediment), there still were clear indications of delays in adult reproduction.

ACKNOWLEDGEMENTS

This study was funded by the U.S. Environmental Protection Agency (EPA) Region 10, Office of Puget Sound, on contract to PTI Environmental Services under the direction of Dr. Thomas C. Ginn.

Research was supervised by Dr. D. Michael Johns of PTI Environmental Services at the E.V.S. Consultants' (E.V.S.) laboratory in North Vancouver, B.C. with the assistance of Dr. Peter Chapman and his staff. *Neanthes* juveniles used in this work were provided by Dr. Donald Reish (California State University, Long Beach).

REFERENCES

Johns, D.M., T.C. Ginn, and D.J. Reish. 1990. Protocol for juvenile *Neanthes* bioassay. Prepared for Puget Sound Estuary Program, Region 10. PTI Environmental Services, Bellvue, WA USA EPA No. 910/90-011.

A TOXICOLOGICAL PERSPECTIVE ON ECOSYSTEM CHARACTERISTICS TO TRACK SUSTAINABLE DEVELOPMENT

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ABSTRACT

"Ecosystem Health," an emerging science paralleling human and veterinary medicine, has as its goals the systematic diagnosis and treatment of stressed ecosystems. Ecosystems are stressed by physical factors such as boat traffic, biological factors such as introduction of an exotic species, and chemical factors such as pH change. Even if these classes of stressors affect the same trophic levels, the resulting ecosystem disease states have different etiologies because the stress is introduced and propagated by different mechanisms.

This paper presents a toxicological perspective on ecosystem sustainability. I discuss how classical toxicological concepts have to be modified when the experimental unit is an ecosystem. When exposures are high, effects are acute and are often measurable (e.g., fish kill). However, when exposures are low and chronic, effects are often hard to separate from the background. Evidence of high risk for lack of sustainable development is exceedence of ecosystem "threshold criteria."

Key Words: Ecosystem Health Sustainable development Ecosystem toxicology Ecosystem risk Ecosystem threshold

INTRODUCTION

In Title III of the Clean Air Act Amendments of 1990, the U.S. Congress identifies an initial list of 191 air toxics whose emissions must be regulated because they "...are known to cause or may reasonably be anticipated to cause...**adverse environmental effects**..." As defined by Congress:

> [T]he term "adverse environmental effects" means any threat of significant adverse effects, which may reasonably be anticipated, to wildlife, aquatic life, or other natural resources including disruption of local ecosystems, impacts on populations of endangered or threatened species, significant degradation of environmental quality over broad areas, or other comparable effects.

With this definition, Congress has issued a mandate for eliminating toxicological impacts to ensure sustainable ecosystems. As commonly defined, an "ecosystem" is an interacting system of living and non-living components in the environment. A "sustainable ecosystem" is one which is adaptive: it can carry or withstand stresses, is prolonged, and is nourished.

Chemicals affect sustainability by their effects on ecosystem processes. The study of toxicologic responses of ecosystems to contaminants has been termed "ecoepidemiology" by Coulston and Korte (Bro-Rasmussen and Lokke 1984). Ecoepidemiological studies are concerned with describing effects, identifying causes, and determining links and pathways in disease processes affecting populations, communities, and ecosystems. Ecoepidemiological analyses involve ecotoxicological evaluations of many types of test systems which are integrated with other data to provide an assessment of the expected damage to ecosystems. This paper examines parallelisms between classical and ecosystem toxicology.

HOW DO CLASSICAL AND ECOSYSTEM TOXICOLOGY DIFFER?

Classical toxicology uses laboratory studies to examine the effects of chemicals on the individual organisms, organ systems, organs, tissues, or cells used in the test and extrapolate these results to populations and other species, usually humans. Classical toxicology can be viewed from at least three perspectives. One perspective focuses on organ systems and target organs, such as the organs of the circulatory, nervous, and respiratory systems. Another perspective emphasizes effects on physiologic functions such as respiration. A third perspective is endpoints. The left column of Table 1 defines several classical endpoints (NRC 1981). The right column is an initial attempt to identify a corresponding ecotoxicological endpoint; some of these are expanded in Table 2.

Ecosystem toxicology is limited by the database available for species of interest. Thus, the Registry of Toxic Effects of Chemical Substances (RTECS) gives data for birds, cat, cattle, horse, chicken, dog, goat, sheep, duck, frog, gerbil, guinea pig, hamster, human, monkey, mouse, pig, pigeon, laboratory quail, rabbit, rat, squirrel, toad, and turkey. Most of the data is for a few species (mouse, rat). Toxicity data for aquatic species are in the database AQUIRE. For the 234 Illinois aquatic species (146 genera) in this database,

TABLE 1. Classical Endpoints of Toxic Effects

(1) TIME SCALES (Hierarchical order):

Acute toxicity	
The adverse effects occurring within a short time of administration of a single dose of a substance or multiple doses given within 24 hours.	The definite structural change (e.g., alteration of diversity or loss of species) occurring within a short time after a single or multiple exposures within 24 hours.
Subacute toxicity (14 day)	
The adverse effects resulting from daily doses administered for a period of 14 days. The observation period is the exposure period.	The adverse functional changes (e.g., decreased productivity, increase or decrease in biomass) resulting from repeated doses over a relatively short period, e.g., arbitrarily, 14 days.)
Subchronic toxicity (90 day)	
Subchronic studies are designed to examine the adverse effects resulting from daily exposure over a portion (<10%) of the average life span of an experimental animal. The observation period is the exposure period plus, e.g., 28 days. These studies provide information on the cumulative toxicity of a substance on target organs and on physiologic and metabolic tolerance of a compound at low-dose prolonged exposure.	The adverse effects to spatial structure and species gradients resulting from repeated exposure for a portion of the lifetime. Many species of plants and animals live decades. Since a redwood lives for 3000 years, a 300 year exposure could be considered "subchronic" by a parallel definition. A functional definition is required: "Subchronic exposure is exposure occurring during <10% of a species' normal mating cycle. For insects the exposure might be a few days, for plants 1 or a few months.
Chronic toxicity (> 90 days to years)	
Chronic toxicity tests are long-term studies carried out over a significant portion of the test animal's lifespan. For the rat, chronic toxicity tests are studies longer than 3 months (i.e., >10% of the life span).	Chronic toxicity are the adverse effects resulting from repeated exposures during a significant portion of a species' life. Chronic toxicity is expected to alter structure and function, and result in low numbers of species, possibly low numbers of organisms, and high variance. (For example, addition of deoxygenating wastes to a stream reduces diversity.)

TABLE 1. Classical Endpoints of Toxic Effects (continued)

(2) ENDPOINTS (Semihierarchical order):	
Mutagenicity	Loss of rare and endangered species. Accelerated development of disease resistance in insects and bacteria.
Teratogenicity (A property of a chemical which causes permanent structural or functional abnormalities during the period of embryonic development.)	Altered succession Selenium-caused collapse of higher trophic species at Kesterson, CA (birds; higher level aquatic species; see Zahn (1986)).
Reproductive toxicity	Lowered reproductive success DDT effect on birds.
Carcinogenicity (A property of a chemical which causes the development of neoplastic lesions during or after exposure to various doses of a test substance by an appropriate route of administration.) (Cancer can be viewed as a loss of, or incorrect operation of, cellular feedback mechanisms.)	A property of a chemical which causes feedback mechanisms to operate inappropriately. Expressions of this include decreased population fitness, adverse changes in age-class structure and predator-prey interactions.
(3) TOXICITY, OTHER (Random order):	
Acute skin and eye irritation (The skin and eye are barriers between an organism's internal organs and the environment. Acute irritation can be viewed as reduction in the effectiveness of this barrier.)	Reductions in ecotones. An ecotone is a transition between two or more diverse communities. (Decreased barrier protection, such as reduced thickness of layers of dead vegetation protecting soil organisms.
Allergic sensitization (Acute hypersensitization or allergy is a pathological state resulting from prior sensitization to a specific molecule or structurally related compound.)	Prior exposure reduces the ability of the ecosystem to withstand subsequent exposures. That is, properties such as resilience, resistance, inertia, and elasticity are decreased.
Neurotoxicity (Neurotoxicity is basically a poisoning ofthe channels involved in information transfer.)	Reduction in system connectivity, i.e., blockages in fluxes. Fire Suppression impedes nutrient flow in forests and grasslands.
Hepatotoxicity (Detoxification.)	Capacity of system to store and detoxify.

there is acute data for 895 compounds (Ross et al. 1986). Only 310 are represented by more than 1 species. Sparks (1989) has remarked:

> "Relatively few species of fish have been commonly used as test organisms in laboratory bioassays. The most commonly used freshwater species include fathead minnows (*Pimephales promelas*), bluegill sunfish (*Lepomis macrochirus*), channel catfish (*Ictalurus punctatus*), goldfish (*Carassius auratus*), common carp (*Cyprinus carpio*), and rainbow trout (*Salmo gairdneri*).
>
> "Few toxicity data exists for the majority of Illinois fishes, including those which are likely to be most sensitive, based on what is known of their habitat preferences and historical changes in their distribution patterns. Such groups of fishes include the darters (Family *Percidae*), redhorse suckers (genus *Moxostoma*), madtom catfishes (genus *Noturus*), and some species of minnows (Family *cyprinidae*). There are 13 endangered and 15 threatened fishes in Illinois whose water quality requirements are largely unknown and will probably not be determined because toxicity testing necessarily involves the death or impairment of hundreds of individual organisms. Of the 199 [fish] species which have occurred in Illinois, toxicity data exist for 84 (Ross et al. 1986), but except for the "standard" bioassay species mentioned above, the data are incomplete, usually consisting of a few chemicals or factors tested under a narrow range of environmental conditions in one laboratory."

It is similarly instructive to examine the universe of phytotoxicity data on terrestrial vascular plants. Fletcher and colleagues (1988) reported that the database PHYTOTOX contained records on 1569 plant species from 682 genera and 147 families.

> "Although this is probably the broadest spectrum of plant taxa ever to be considered in a toxicity database, the numbers of taxa are still quite restricted, inasmuch as the plant kingdom comprises approximately 250000 species, 12000 genera and 300 families... In the database, 42% of the records deal with only 20 plant species, of which 19 are agronomic plants... Approximately 25% of the data deals with only 20 of the 5000 compounds present in the database...In contrast to the abundance of data on the common plant hormones and herbicides, there is negligible or no information on most of the human carcinogens and toxic waste compounds that are of primary interest to the U.S. Environmental Protection Agency. For example, of the 114 organic chemicals included on the EPA list of priority pollutants, PHYTOTOX contains records for only 40 [including 12 pesticides]."

Classical toxicology is usually defined in terms of an exposure protocol which does not consider the magnitude and duration of the response: "Acute toxicity causes death or extreme physiological disorders to organisms immediately or shortly following exposure to the contaminant. Chronic toxicity involves long-term effects of small doses of a contaminant and their cumulative effects over time. These effects may lead to death of the organism or disruption of such vital functions as reproduction" (USEPA 1989, p. 23). In contrast, medical usage recognizes that acute or chronic exposure can have lethal or sublethal effects. Baillière's medical dictionary (Blood and Studdert 1988) defines "acute" as "having severe signs and a short course of 12-24 h." "Chronic" means "persisting for a long time; the period is undefined and varies with the circumstances. It also has the sense of the disease showing little change or very slow progression over a long period." Ecosystem Health is a medical science so the classification of ecotoxicities must include the exposure duration and the response duration and magnitude. A preliminary classification is attempted in Table 2.

Ecotoxicology studies the effects of chemicals on the structure and function of biotic communities and on their interactions with abiotic components. The "structure of an ecosystem is defined by the abundance and biomass of all populations and their spatial, taxonomic and trophic organization" (Sheehan 1984). Measuring the stability of an ecosystem is therefore a diagnostic goal of Ecosystem Health studies. Attaining and ensuring a sustainable ecosystem requires attaining and ensuring ecosystem stability. Table 3 defines terms used to describe ecosystem stability.

Ecosystem effects of chemicals (NRC 1981; Sheehan 1984) generally are not included in classical toxicology. The outstanding need of ecosystem toxicology is to evaluate subtle, complex effects occurring concurrently in multiple species that are not normally studied by classical toxicologists. The relevant responses, such as changes in species diversity, are difficult to define clearly and unambiguously. Generally, these responses have high measurement variance and unknown accuracy. While many classical responses are components of an ecosystem "health" assessment, they are seldom endpoints *per se*. System complexity and data scarcity limit the ecotoxicologist's ability to predict the

TABLE 2. Comparison of Classical with Ecotoxicology

CLASSICAL	ECOSYSTEM
Animal Response	**Ecosystem Response**
Acute Exposure, Acute Response	
Example: Strychnine poisoning Exposure to NH_3 or HCl gas	Example: Fish Kill due to NH_3 runoff Immediate effects of oil spill, e.g., suffocation
Acute Exposure, Chronic Response	
No standard definition.	The definite structural change occurring over a long time, (months to years) after single or multiple exposures within 24 hours.
Example: Organophosphorus-induced delayed neuropathy (OPIDN)	Example: Long term effects of oil spill
Triorthocresol phosphate (TOCP) in bootleg alcohol caused a massive outbreak of neuropathy in the late 1920's. The paresis of upper and lower extremities was known as *ginger jake paralysis.*	Loss of critical habitat, delayed toxicity and photo-induced toxicity, bioaccumulation of toxic residues.
Radiation exposure	Raditation Exposure (blast)
Loss of organ system function such as red blood cell production and immune system; germ cell effects (mutations).	Loss of species and trophic levels; germ cell effects (mutations, evolutionary "fitness" of species.)

TABLE 2. Comparison of Classical with Ecotoxicology (continued)

Chronic Exposure, Acute Response

Copper-molybdenum deficiency in sheep produces an acute syndrome which suddenly occurs after chronic dietary exposure to an excessive copper to molybdenum ratio (>10:1). (The proper ratio is 6:1 and 8-10 ppm of copper in diet.)	Acid rain leaches minerals from thin Adirondack soils thereby permanently changing soil pH. Runoff enters lackes where metals decrease water pH, which in turn increases metal ion solubility. Death ensues from multiple causes.
Molybdenum toxicosis-copper deficiency Bovine syndrome: Morbidity may approach 80%. Diarrhea with gas bubbles. Emaciation. Decreased milk production. Decreased fertility and lameness. Anaemia. Chronically–bone fractures. Sheep syndrome: Lambs–ataxia, blindness.	Genetic change due to pollution a) The classic example is a genotypic color change (from light to dark in British moths exposed to trees darkened by coal-fire smoke. b) Selenium case in Table 1.

Idiosyncratic

Abnormal sensitivity, or lack of sensitivity, to exposure by some members of population.	"Most sensitive" species. The species mean LC50 for benzene is 5,300 μg/ℓ for rainbow trout and 380000 μg/ℓ for *Daphnia magna*. Some *D. magna* populations had an LC50 of 620000 μ/ℓ.

TABLE 2. Comparison of Classical with Ecotoxicology (continued)

TOXIC MODE OF ACTION	**ECOTOXIC MODE OF ACTION**
Mutagenesis	Genotypic and phenotypic diversity.
Teratogenesis	Reproductive effects of individual, species.
Carcinogenesis	Fitness (Totter 1981) gives evolutionary basis of carcinogenesis.)
Allergic sensitization	---------
SUBSYSTEM (Organ system)	**SUBSYSTEM (Trophic Level)**
(Parallelism not implied)	
Neuromuscular Hepatic Reproductive	Hierarchial structure altered or destroyed (see Allen et al. 1984). Abundance and Biomass Reduction in Population Size and Extinction Loss of species with unique functions Species richness Community Composition-and Species Dominance Species lists Indicator species Biological indices Dominance patterns Species Diversity/Similarity Spatial Structure Stability (see Table 3)

magnitude and significance of toxic effects of chemicals on ecosystems. Models of ecosystem effects have been proposed, but most have substantial data requirements and model results are difficult to interpret. Another limiting factor is that ecosystem toxicology must use limited manipulation and pseudoreplication of the ecosystem in place of the highly controlled and replicated experiments used in classical toxicology.

Several ecosystem components are "critical" to maintenance of the ecosystem. These characteristics are discussed in a report of the National Academy of Science (1981). Herricks and schaeffer (1987) identified 8 *measurable* characteristics likely to be affected by chemicals and gave 44 measures appropriate to individuals, populations, ecosystem biological components, and major abiotic elements (Schaeffer et al. 1988). The 8 critical characteristics (which are not of equal importance in every ecosystem) and classical toxicology analogs are given in Table 4. For example, in the same fashion that respiratory and basal metabolic rates provide system information to the classical toxicologist, photosynthesis and nutrient cycling rates provide system information to the ecotoxicologist.

Table 4 also gives trends that may be expected in ecosystems upon the advent of stress (Odum 1985). Not all of these effects are likely to occur in any particular, newly stressed ecosystem. The intensity and duration of the stress are both important in this respect. For example, stress in many freshwater aquatic ecosystems will cause large changes in species composition (such as in benthos and phytoplankton) before changes take place in ecological functions such as respiration or primary production. "In such a case the toxic threshold for sensitive taxa occurs at a lower intensity of stress than does the toxic threshold for function, which may be carried out at a community level irrespective of the species composition. Therefore, changes in species composition and demographics would be expected to be relatively sensitive indicators of the initial ecosystem response to the advent of a low or moderate intensity of stress" (Freedman 1989, p. 320). However, in many terrestrial ecosystems, such as forests (Schindler 1988) and grasslands (Schaeffer 1989), the reverse may be found. "[A] moderate intensity of stress may affect functional attributes such as photosynthesis, respiration, and nutrient cycling well before there is mortality of individual plants or elimination of species" (Freedman 1989 p. 320)

THE EMERGING SCIENCE OF ECOSYSTEM HEALTH

The bases of the new science of Ecosystem Health are: determination of the existence of stress; systematic study of the effects of stress on ecosystem structure and function; and relief of stress to allow normal ecosystem functioning. The science of Ecosystem Health is analogous in its goals and methods to those of the human and animal health sciences (Table 5). Evolution of the science of Ecosystem Health will lead to development and use of standardized diagnostic methods as "markers" of *effect*, specialized language, catalog of diseases, ecosystem specialists, and in the fullness of time, treatment methodologies for ill ecosystems. The conceptual stage for development of the science of Ecosystem Health has been set thus (Schaeffer et al. 1988).

> "The classical definition of an ecosystem couples interacting living organisms and nonliving components of the environment to form one physical system and grew from the recognition that definable and describable units existed in nature. Today we approach the protection, management, or restoration of natural environmental conditions using conditions which may or may not be ecologically relevant. Although the maintenance of ecosystem integrity requires more than single species management or protection, environmental protection is constrained by analyses which. incompletely integrate ecosystem complexity and provide results with only limited ecological relevance. It is difficult to extrapolate laboratory testing to actual ecosystem effect...[M]easures of human or nonhuman animal health, and the clinical analysis of factors which contribute to a definition of a state of health, provide useful analogs to the problems faced by environmental managers attempting to maintain the integrity of ecosystems."
>
> "Environmental decision making is limited, and is often fundamentally flawed, because of the inability to relate data in an ecosystem context. The issue is not always that insufficient or inadequate data are available...[but] that scientists and engineers fail to extract all the information possible from existing data sets. The lack of data may be the excuse for the absence of decisions.-..[A] more pertinent issue is the lack of a common set of analysis approaches...which translate data on the complexity of ecosystems into simple and understandable information about state or condition which will support decision making."

TABLE 4. Classical Toxicological Analogs of Ecosystem Critical Characteristics

Ecosystem "Critical" Characteristic and trends in stressed ecosystem[1]	**Classical Toxicological Analog**
I. Habitat is suitable for maintaining diversity and reproduction of organisms at evolutionarily acceptable levels. Stressed ecosystem trends include: Ecosystem becomes more open. Successional trends reverse. Parasitism and other negative interactions increase, mutualism and other positive interactions decrease. Sensitive genotypes are replaced by more tolerant genotypes.	Habitat is suitable for maintenance of test individuals.
II. Phenotypic and genotypic diversity exists and is maintained among organisms. Stressed ecosystem trends include: Large species lost. Decrease in lifespans of organisms or parts (e. g., leaves). Species diversity decreases and dominance increases.	Usually, phenotypic and genotypic similarity of test organisms is desired and maintained.
III. A robust food chain supporting the desired biota exists. Stressed ecosystem trends include: Proportion of r-strategists increases. Food chains shorten.	Metabolism
IV. An adequate nutrient pool for desired organisms exists. Stressed ecosystem trends include: Nutrient loss increases (i.e., system becomes more "leaky").	Metabolic reserves
V. Nutrient cycling is adequate to perpetuate the ecosystem. Stressed ecosystem trends include: Food chains shorten. Maintenance to biomass structure (P/B and R/B) ratios increase. Nutrient turnover time increases. Horizontal transport increases and vertical cycling of nutrients decreases.	Catabolism

TABLE 4. Classical Toxicological Analogs of Ecosystem Critical Characteristics. (Continued)

VI. Energy flux is adequate to maintain the trophic structure. Stressed ecosystem trends include: Ecosystem becomes more open and internal cycling is reduced. Efficiency of resource use decreases. Importance of auxiliary energy increases. Community respiration decreases. Production/respiration becomes unbalanced (i.e., P/R < > 1).	Respiration, metabolism, catabolism are adequate to maintain normal growth rate.
VII. Feedback mechanisms for damping undesirable oscillations are in place and adequate. Stressed ecosystem trends include: Redundancy of parallel processes declines. Successional trends reverse to earlier stages.	Feedback mechanisms, such as hormone levels are in place and adequate.
VIII. An adequate capacity to temper toxic effects, including the capacity to decompose, transfer, chelate or bind anthropogenic inputs to a degree that they are no longer toxic within the system. Stressed ecosystem trends include: Efficiency of resource use drops. Ecosystem becomes more open so that inputs and outputs become more important as internal cycling is reduced. Size of organisms decreases. Species diversity decreases.	Detoxification mechanisms, including metabolism, binding and excretion have adequate capacity and rate.

[1] Ecosystem trends adapted by author from Odum (1985).

MARKERS OF EFFECT: ECOSYSTEM THRESHOLD CRITERIA

The issue of sustainability can be viewed as the maintenance of ecosystem stability. An ecosystem is stable if, and only if, the variables all return to the initial equilibrium following their being perturbed from it (Pimm 1984). An ecosystem is locally stable if this return is known to apply only certainly for small perturbations and globally stable if the system returns from all possible perturbations. Based on classical toxicology we infer that instability occurs when an ecosystem "threshold" is exceeded. Thus, Woodwell (1975) asked:

Is it reasonable to assume that thresholds for effects of disturbance exist in natural ecosystems? Or are all disturbances effective, cumulative, and detrimental to the normal functioning of natural ecosystems?

The question is analogous to the classical question of hazards of ionizing radiation and other toxins. If small exposures in addition to background have no discernible effect, but larger exposures do, then it is common to speak of the maximum exposure that has no effect as a "threshold."

The existence of ecosystem thresholds, real or apparent, are assumed (*de facto*) to provide a practical basis for regulation of a pollutant or other types of stressors, such as vehicle traffic. However, it may be that exact criteria cannot be developed. Thus, based on whole ecosystem radiation studies, Woodwell (1975) concluded that:

> "If we seek thresholds along [an exposure] gradient, we can find them, but not at the ecosystem level. The thresholds are for survival of species, for the elimination of trees, for the invasion of adventives, for measurable effects on growth. If we measure effects on growth, we find that greater refinement in analysis shows effects at lower concentrations. Any threshold is arbitrary."

> "...[I]f thresholds exist for effects of disturbance, they are few, and they lie at exposure levels below those at which they can be resolved by most current studies of ecosystems per se."

A marker of effect is evidence that an (apparent or real) ecosystem threshold criterion has been exceeded. A preliminary definition of a what Schaeffer and Cox (1990) term a "functional" ecosystem threshold criterion is:

> "Any condition (internal or external to the system) which, when exceeded, increases the adverse risk to maintenance of the ecological system."

Systematic development of functional ecosystem threshold criteria is a significant new research area which encompasses and integrates the judicial, political, social, and economic disciplines, and the biological, chemical, and physical sciences.

Scientific criteria are conveniently classified into three categories. Biological threshold criteria encompass a wide range of species or community responses. For example, Karr's (1981) Index of Biotic Integrity is computed using 11 characteristics of the midwest fish community. Threshold criteria can be developed using the *physical* properties of the abiotic components of the environment. For example, the Predicted Index of Biotic Integrity (PIBI) (Hite and Bertrand 1989) uses stream physical characteristics to predict the maximum possible IBI score. Criteria may also be based on the *chemical* properties of the abiotic components. Changes in energy flux relate the abiotic to the biotic components, and could be a basis for threshold criteria. Scientific criteria can be developed through the formal process of hypothesis testing (e.g., IBI), modeling (e.g., PIBI), or consensus (e.g., Delphi) procedures (Brown et al. 1970).

Development of scientific criteria is a complex undertaking which involves, initially, identification of appropriate biotic measures and, subsequently, controlled testing and refinement. A comprehensive effort will require identification of existing criteria, measures which could be readily modified to provide initial criteria, and a process/data requirements for developing criteria. Each type of ecosystem has unique species, abiotic components, and interactions which must be dealt with by experts in the ecology of that type of ecosystem. For example, while a threshold for "sufficient nutrient cycling" is applicable to all types of ecosystems, the criterion value and method for its measurement can be different for aquatic, prairie, and forest ecological systems.

RISK ASSESSMENT FOR ECOSYSTEMS

Most risk methods for chemicals focus on single chemicals and human cancer. Comprehensive methods are not available for assessing the toxicity/risks for mixtures, time-ranging concentrations (however, see Mancini 1983) and ecosystems. A natural species may be simultaneously subjected to several stressors. In a population subjected to substantial pressure, the stability will be low and the individual risk factors can interact synergistically to produce a total risk which is much greater than the simple sum of the individual risks. Barnthouse and colleagues (1990) demonstrated this for fisheries populations using elegant, powerful, risk models that combine laboratory toxicity test data with life history data. For example, models showed that because of differences in life history, the heavily exploited species (Chesapeake Bay striped bass) had a much lower capacity to sustain contaminant-induced mortality (from trifluralin) than did Gulf menhaden. Their results suggest that consideration of life history may be important for site-specific assessments, but that the substantial differences in uncertainty associated with different types of test data are of much greater concern for screening-level assessments.

Species show varying sensitivities and responses to toxic substances, so it is generally not possible to deduce the response of an ecosystem from the response of a single species. Yet, the predictive relationships which have been developed among many species suggest that relationships can be developed which use the response of a single species (or a small number of species) to predict the response of an ecosystem. According to published criteria, a species has ecological importance if it: 1) provides more than 20% of total biomass at a given trophic level; 2) provides a primary food supply (more than 30%) to an organism with economic or social value; 3) provides a secondary food supply (>20%) to an organism with economic or social value; 4) is a primary producer or primary producing group that provides more than 20% of the food base for primary consumption; 5) is a dominant predator than consumes more than 15% of a given species: or 6) provides important habitat space for other organisms.

Because of uncertainties in environmental monitoring (exposure) and biological (response) data, susceptibilities of individuals and species (Storer 1975; Totter and Weinberg 1977), and the like, the exposure and the dose-response curves, and hence the risk curve, cannot be accurately determined (*supra* Barnthouse et al. (1990)). However, if uncertainties are small enough to permit at least confidence region estimates for the component curves, then probability convolution methods (Kaplan 1981; Springer 1979), can be used to compute simultaneous confidence regions for the risk frequency curve (Clarke and Schaeffer 1990). Point estimates for the various risk statistics of interest are thus replaced by approximate numerical confidence interval estimates for these statistics.

CONCLUSION

This paper has attempted an initial comparison between those aspects of classical toxicology and the emerging field of ecosystem toxicology related to the development, characterization, and maintenance of sustainable ecosystems. Although many components of ecosystem toxicology can be analogized to elements of classical toxicology, the ecosystem toxicologist must guard against oversimplification. There are at least eight differences which must be emphasized.

(1) The multispecies structure of an ecosystem cannot be fully analogized to an individual or a laboratory colony.

(2) Genotoxic effects in a population affect a species' evolutionary fitness which is not analogous to survival of an individual.

(3) Energy and nutrient flows and balances in an ecosystem include the abiotic components and the biotic components at numerous trophic levels, and these are not always identifiable or measurable.

(4) Environmental factors can cause large geographic and annual fluctuations (coefficient of variation > 30%) in population measures such as numbers of organisms, age class distribution and diversity. Classical toxicologists accustomed to relatively low variability in inbred animals and statistical criteria of $\alpha = 0.05$, power = 0.8 to judge treatment differences, will have to adjust to the use of more liberal criteria, such as $\alpha = 0.1$ or $\alpha = 0.2$, power = 0.6., for judging ecosystem differences.

(5) The ecotoxicologist works in complex systems which can have multiple mechanisms for tempering toxic effects. Consequently, an effect which is "significant" to the classical toxicologist may not be found by the ecotoxicologist. For example, because nutrient recycling was very rapid in a prairie ecosystem exposed to both chemicals and heavy vehicle traffic, a field study did not find exposure effects although significant mortality was found in plants exposed in the laboratory to these chemicals (Schaeffer et al. 1990).

(6) Classical toxicology relies on replication whereas ecosystem toxicology is limited, at best, to pseudoreplication (Hurlbert 1984) through use of devices such as limnocorals.

(7) The concept of an "ecosystem threshold criterion" is very new, and until the theoretical concepts needed to develop criteria and measure effect are identified, ecosystem toxicology will continue to emphasize studies of acute effects resulting from high-level acute exposures and not studies of effects resulting from low-level chronic exposures.

(8) The complexity of the ecosystem must never be underestimated or over-simplified. For example, small intermittent streams are often expendable in the human system of values. However, these are important to the sustainability of larger perennial streams because they carry a diverse, annually highly variable, biota which contribute both to an individual species' gene pool and to a community's species pool.

Ecotoxicology is concerned with the assessment of ecosystem risk and with development of methods which will enable resource managers to minimize ecosystem risk. We can (intuitively or quantitatively) express "risk" to an ecosystem as a metric between 0 and 1. If ecosystem risk is low, then ecosystem health is high and, by definition, the ecosystem is sustainable. Being ecosystem risk assessors, ecotoxicologists will therefore play a key role in any effort to maximize ecosystem sustainability.

REFERENCES

Allen, T.F. H., R. V. O'Neill and T. W. Hoekstra. 1984. Inter-level Relations in Ecological Research and Management: Some Working Principles from Hierarchy Theory. General Technical Report RM-110, United States Department of Agriculture, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Barnthouse, L. W., 6. W. Suter II and A. E. Rosen. 1990. Risks of toxic contaminants to exploited fish populations: Influence of life history, data uncertainty and exploitation intensity. Environ. Tox. Chem. 9:297-311.

Blood, D. C. and V. P. Studdert. 1988. Baillière's Comprehensive Veterinary Dictionary. Baillière Tindall: Philadelphia, PA.

Bro-Rasmussen, F. and H. Lokke. 1984. Ecoepidemiology–a casuistic discipline describing ecological disturbances and damages in relation to their specific causes: exemplified by chlorinated phenols and chlorophenoxy acids. Regulat. Toxicol. Pharmacol. 4:391-399.

Brown, R. M., N. I. McClelland, R. A. Denninger and R. G. Tozer. 1970. A water quality index–do we dare? Water Sewage Works 117:339-343.

Clarke A. C. M. and D. J. Schaeffer. 1990. Probabilistic risk curves from linearized exposure and dose-response curves: application to a nuclear radiation hazard. The Environmental Professional 12:114-120.

Fletcher, J. S., F. L. Johnson and J. C. McFarlane. 1988. Database assessment of phytotoxicity data published on terrestrial vascular plants. Environ. Toxicol. Chem. 7:615-622.

Freedman, B. 1989. Environmental Ecology. Academic Press, New York.

Herricks, E. E. and D. J. Schaeffer. 1987. Selection of test systems for ecological analysis. Water Sci. Technol. 19:47-54.

Hite, R. L. and B. A. Bertrand. 1989. Biological Stream Characterization (BSC): A Biological Assessment of Illinois Stream Quality. Special Report No. 13 of the Illinois State Water Plan Task Force. IEPA/WPC/89-275. Illinois Environmental Protection Agency, Division of Water Pollution Control, Springfield, IL.

Holling, C. S. 1973. Resilience and stability of ecological systems. Ann. Rev. Ecol. Syst. 4:1-24.

Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecolog. Mongr. 54(2):187-211.

Kaplan, S. 1981. On the method of discrete probability distributions in risk and reliability calculations: Application to seismic risk assessment. Risk Analysis 1:189-196.

Karr, J. 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6):21-27.

Kerster, H. W., D. J. Schaeffer and K. A. Reinbold. 1988. Assessing Ecosystem Impacts from Simulant and Decontaminant Use. CRDEC-CR-88059. U.S. Army Armament Munitions Chemical Command, Aberdeen Proving Ground, MD.

Mancici, J. L. 1983. A method for calculating effects, on aquatic organisms, of time varying concentrations. Water Res. 17:1355-1362.

NRC. 1981. Testing for Effects of Chemicals in Ecosystems. National Academy Press, Washington, D.C.

NRC. 1984. Toxicity testing: Strategies to Determine Needs and Priorities. National Academy Press, Washington, D.C.

Odum, E. P. 1985. Trends expected in stressed ecosystems. BioScience 35:419-422.

Pimm, S. L. 1984. The complexity and stability of ecosystems. Nature 307(26):321-326.

Ross, P., J. M. Kamin, L. C. Burnett and J. J. Frick. 1986. Summary of Toxicological Data for Aquatic Organisms of Illinois. Parts I, II, III. Publication No. HWRIC RR 009. Hazardous Waste Research and Information Center, University of Illinois, Urbana, IL.

Schaeffer, D. J. and D. K. Cox. 1990. Approaches to establish ecosystem threshold criteria. Ecosystem Health. VIII. In Proceedings of the Ecosystem Health and Environmental Management Workshop. Wye Woods Centre at the Aspen Institute, Maryland, October 28-30, 1990. In press.

Schaeffer, D. J., E. E. Herricks and H. W. Kerster. 1988. Ecosystem Health. I. Measuring Ecosystem Health. Environ. Manage. 12:445-455.

Schaeffer, D. J., T. R. Seastedt, D. J. Gibson, D. C. Hartnett, B. A. D. Hetrick, S. W. James, D. W. Kaufman, A. P. Schwab, E. E. Herricks and E. W. Novak. 1990. Ecosystem Health. V. Use of field bioassessments to select test systems for relevant impact assessments or hazard evaluations. Training lands in tallgrass prairie. Environ. Manage. 14:81-93.

Schindler, D. W. 1988. Effects of acid rain on freshwater ecosystems. Science 239:149-157.

Sheehan, P. J. 1984. Effects on community and ecosystem structure and dynamics and functional changes in ecosystems, pp. 51-145. In P. J. Sheehan, D. R. Miller, G. C. Butler, and P. Bourdeau, eds., Effects of Pollutants at the Ecosystem Level. John Wiley, NY.

Sparks, R. E. 1989. Sparks Comments on Toxics Control. Letter to Mr. Tom Heavisides, Illinois Department of Energy and Natural Resources, June 19. (As quoted in testimony before the Illinois Pollution Control Board: Philo, G., T. Heavisides, and D. Schaeffer. 1989. Analysis of Proposed Revisions to Subtitle C Toxics Control Program: Pollution Control Board Docket R88-21, pp. 6-12 to 6-13.)

Springer, M. D. 1979. The Algebra of Random Variables. John Wiley, NY.

Storer, J. B. 1975. Radiative carcinogenesis, pp. 453-483. In F. Becker, ed., Cancer (vol. 1). Plenum Press, NY.

Totter, J. R. 1981. The origins of cancer: a comparison of two hypotheses. Period. Biol. 83:209-??.

Totter, J. R. and A. M. Weinberg. 1977. Repair and dose-response at low doses. ORAU/IEA(0) -77-11. Oak Ridge Associated Universities, Oak Ridge, TN.

USEPA. 1989. Risk Assessment Guidance for Superfund. Volume II. Environmental Evaluation Manual. EAP 540/1-89/001. Interim Final. U.S. Environmental Protection Agency, Washington, D.C.

Westman, W. E. 1978. Measuring the inertia and resilience of ecosystems. Bioscience 28:705-710.

Woodwell, G. M. 1975. Threshold problems in ecosystems, pp. 9-21. In S. A. Levin, ed., Ecosystem Analysis and Prediction. SIAM, Philadelphia, PA.

Zahn, G. R. 1986. Kesterson Reservoir and Kesterson National Wildlife Refuge, California, USA, history and current problems in management alternatives, pp. 324-329. In R. E. McCabe, ed., Transactions of the North American Wildlife and Natural Resources Conference, No. 51. Resource Management: First Line of Natural Defense; Reno, Nevada, USA, March 21-26, 1986. Wildlife Management Institute, Washington, D.C.

MONITORING FOR SUSTAINABLE DEVELOPMENT EXTENDED SUMMARY

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One of the key concerns regarding sustainable development is developing and implementing monitoring strategies to measure the state of ecosystems, resources, and conditions sustaining human health. Monitoring for sustainable development requires measurement of system attributes. This ecosystem approach involves first establishing monitoring goals consistent with sustainable development, then developing a sampling strategy to meet the monitoring goal.

Most monitoring programs measure only "simple" variables whose behaviour is determined by only a few processes. Examples would be the concentration of phosphorus in water or the concentration of dioxins and furans in fish tissues. Monitoring for sustainable development must measure also "complex" system attributes with multiple functional linkages to many other ecosystem variables. Such attributes include trophic structure, nutrient flow, productivity of trophic levels and of ecosystems, energy flow, ecosystem state, and resilience. Ecosystem state is a description of a characteristic qualitative functioning mode of an ecosystem. Major functional pathways, key and dominant species, and general mode and rate of production remain relatively constant. Change to different ecosystem states implies a qualitative or major quantitative jump to different ecosystem characteristics. Resilience is the ability of a system to oscillate but remain in the same qualitative state (ecosystem state in this discussion). These system attributes are both more abstract and more complex. In many cases, effective measures or indicators of these attributes have yet to be developed. For example, although total phosphorus at spring overturn has been found to be a useful indicator of trophic state for lakes that stratify, we do not have a similarly predictive measure of trophic state for flowing or marine waters. Useful indicators or measures of resilience are still to be developed as well.

Monitoring goals for sustainable development focus on gathering information that will anticipate changes in ecosystem state, a difficult goal considering our lack of understanding of the alternate states in which an ecosystem might exist. An ecosystem might change from its current state to a different, yet sustainable, ecosystem. One important goal of monitoring for sustainable development is to estimate the probability and risk of such a state change.

In this perspective the monitoring program should, at each "sampling," provide *risk status*. Implementing monitoring for *risk status* will require measuring ecosystem state, ecosystem resilience, and probability of a state change. Measurement and development of indicators of ecosystem state, resilience, and probability of a state change requires an extensive and intensive development effort.

In addition to knowing the *risk status* it is necessary to evaluate the consequences of state changes. Predicting the consequences of state changes involves evaluation of alternate states. Managers must be able to evaluate the social/environmental desirability of potential alternate states versus the current state of the ecosystem. Evaluation of alternate states is difficult and always entails uncertainty. Some methods, such as modelling, microcosm and field experimentation, and case studies of managed ecosystems, are available as starting points to predict alternate states. In some cases, alternate states could have greater overall value, including both monetary and non-monetary considerations, to humans. This evaluation is an opportunity for application of decision support systems and public input processes.

WATER QUALITY GUIDELINES AND OBJECTIVES FOR SUSTAINABLE DEVELOPMENT

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ABSTRACT

To realize the goals of sustainable development and bring environmental factors into the mainstream of decision making, we must develop the scientific, environmental framework necessary to move sustainable development from a vaguely defined political philosophy to a practical, operational concept. Water quality guidelines and objectives define realistic and scientifically defensible measures and goals of environmental sustainability that can be incorporated into the decision making process alongside conventional economic and social criteria. Water quality guidelines and objectives are now widely used across Canada. The "goals and yardstick" value has potentially broad applications to the sustainable development of water resources; i.e. environmental impact assessment, site rehabilitation, state of the environment reporting, and the development of standards for aquatic ecosystem protection. With reference to Canadian programs, we discuss current trends in the development and application of water quality objectives to sustaining aquatic ecosystems. Topics include the development of water quality objectives for transboundary water management, the development of ecosystem objectives for Lake Ontario and the importance of public participation, and the move from traditional "end-of-pipe" approaches to environmental protection and regulation to approaches incorporating specific water quality and ecosystem objectives for receiving waters.

INTRODUCTION

Sustainable Development

Since the World Commission on Environment and Development (Brundtland Commission) released its landmark report in 1987, the concept of sustainable development has pervaded social, economic and ecological theory in Canada and set a new course for the way we view the environment and our relationship to it. Sustainable development, perhaps more accurately referred to as "environmentally sustainable economic development", has been variously described as "development which meets the needs of the present without compromising the ability of future generations to meet their own needs" (World Commission on Environment and Development) and "development which ensures that the utilization of resources and the environment today does not damage prospects for their use by future generations" (National Task Force on Environment and Economy). Though definitions may vary, they are united by a common ethic – the right of future generations to the resources of the planet – and are based on the premise that we can have economic development over the long-run only if we maintain long-term health and integrity of the environment.

The Challenge of Sustainability – From Rhetoric to Reality

"... the beguiling simplicity of the term sustainability has placed it in grave danger of becoming so totally accepted politically, that it will become meaningless; languishing as a "good idea" which cannot be put into practice..."
(O'Riordan, 1988)

Though the principles of sustainability may be clear, precise definitions and measures of sustainability remain elusive (e.g. Shearman, 1990). The current challenge is to move from a vaguely defined political philosophy to an operational, practical concept or strategy (Slater, 1988). We must provide credible goals and measures of environmental sustainability so that environmental factors can be shifted into the mainstream of decision-making at all levels as the basis for development of environmentally sustainable policies, programs and projects (Environment Canada, 1990).

Water Quality Guidelines and Objectives

Water quality guidelines are an important component in developing an operational framework for sustainable development. They provide scientifically defensible, precisely defined numerical limits or narrative statements for sustaining aquatic environmental quality and uses that can be incorporated into the decision making process alongside conventional economic and social criteria. Water quality guidelines are based on an integration of environment/economy data, such as environmental response to potential human-induced stresses (e. g. ecotoxicology) and water uses; uniting science, technology and public values (Hamner, 1989) to define realistic and scientifically defensible goals and measures for sustainability.

The "goals and yardstick" function has potentially broad application to such areas as impact assessment, state of the environment reporting and environmental management and rehabilitation. *Water quality guidelines* can be used directly to assess water quality issues and concerns but they also serve as the technical basis for the development of *water quality objectives* – numerical concentrations or narrative statement established to support and protect the designated uses of water at a specified site (CCREM 1987b). Water quality objectives, in turn, provide a framework for the development of *water quality standards* – objectives that are recognized in enforceable environmental control laws of a level of government. Development and harmonization of standards and legislation for environmental protection has been identified as a primary goal for sustainable development in Canada (CCREM, 1987a). Figure 1 indicates the theoretical placement of water quality guidelines, objectives and standards in E. W. Manning's "building blocks" of sustainable development (Environment Canada, 1990).

The nature and role of water quality guidelines and objectives in the sustainable development of Canada's water resources will be discussed with reference to several Canadian examples.

CANADIAN WATER QUALITY GUIDELINES

Canadian provinces began to develop water quality objectives in the 1960s and 70s. In 1987, the CCME (formerly CCREM) published the Canadian Water Quality Guidelines to provide a nationally consistent scientific basis for developing water quality objectives. The Canadian Water Quality Guidelines synthesize a broad base of ecotoxicological, environmental and human use information to define benchmark values based on the degree to which aquatic systems can withstand or sustain contamination or other perturbations without impairment of important uses or biotic integrity (aquatic life). Guidelines specify water quality (determined by the kinds and amounts of matter which is dissolved and suspended in water such as bacteria, pesticides and metals) for single uses such as livestock watering and recreation. Specifying parameters for distinct uses (including aquatic life), maintains maximum flexibility in application of the guidelines to development of site-specific objectives which define the water quality necessary to sustain all existing and intended uses at a particular site. This guidelines/objectives approach provides a clear strategy for dealing with the multi-functional nature of water resources and provides realistic, integrated measures or goals for sustaining aquatic resources that can be tailored to specific uses, societal values, and environmental characteristics of a site.

Canadian Water Quality Guidelines (CCREM 1987b) are now widely used across Canada and address a range of potential uses (recreation, irrigation, livestock watering, municipal, freshwater aquatic life). The guidelines contain recommendations for chemical, physical, radiological and biological parameters necessary to protect and enhance these uses, including the forms and fate of the parameters. For example, more than 50 parameters have been addressed in the formulation of guidelines for freshwater aquatic life.

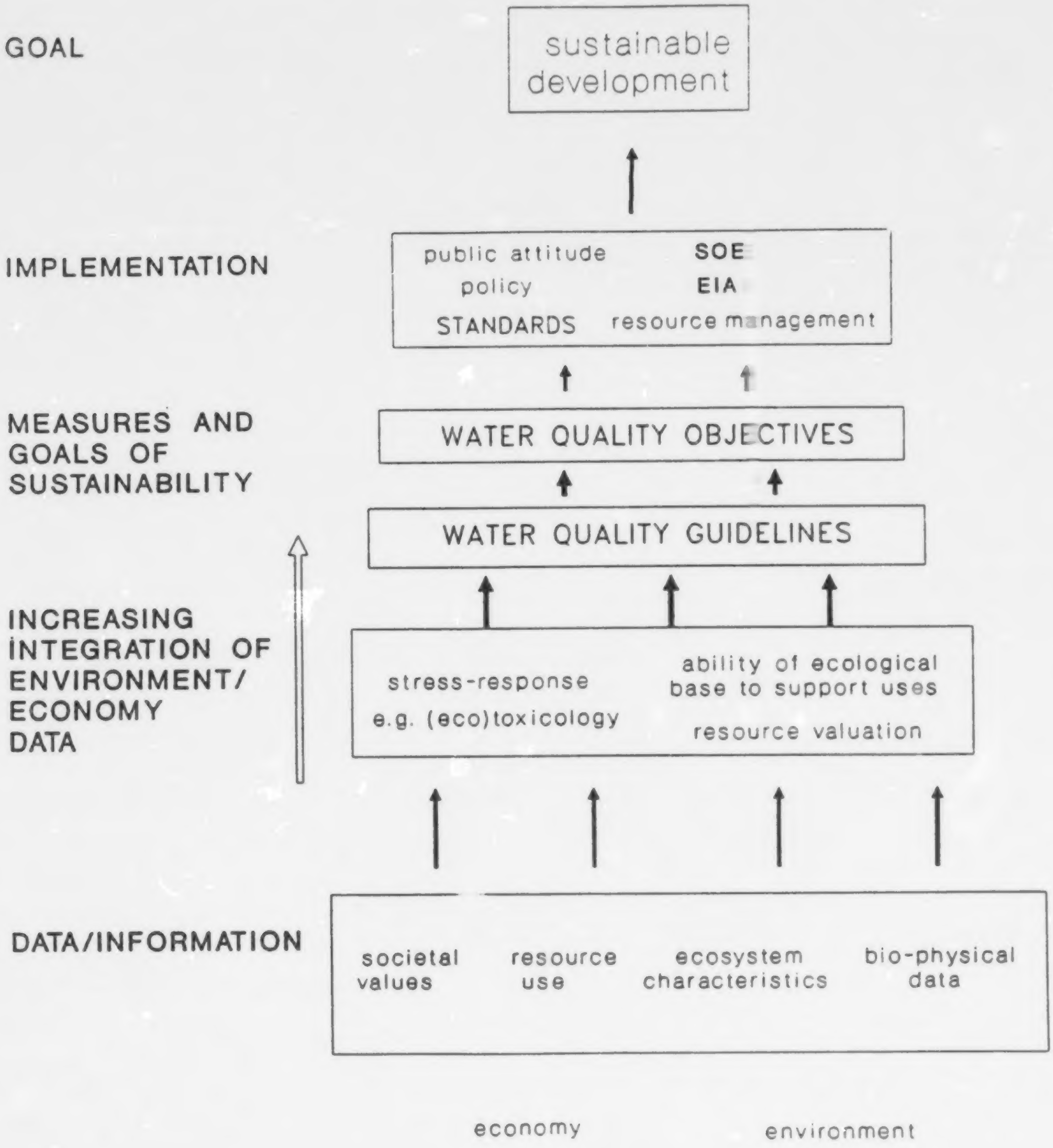
EMERGING TRENDS IN DEVELOPMENT AND APPLICATION OF WATER QUALITY OBJECTIVES

Water Quality Objectives for Transboundary Water Management

Water quality objectives have played a key role in transboundary water management in Canada. Objectives were set for the Red River in 1969, the Great Lakes in 1972 and 1978, and the Poplar River in 1981. Since 1969, the Prairie Provinces Water Board (a federal-provincial board which coordinates interprovincial water management for the prairie provinces) has been working to develop water quality objectives for effective interprovincial water quality management. In 1973, the PPWB Water Quality Objectives were adopted as a first step in the process of establishing an interprovincial water quality management strategy. These objectives were defined as water quality "aims or goals" toward which to strive in order to ensure uses and quality are maintained (sustained) in light of existing and anticipated stresses on the system (e.g. hydro-electric development) (PPWB, 1989).

The PPWB incorporates water quality objectives as a cornerstone in delivering its water quality mandate (PPWB, 1990) which "encourages the protection, restoration, and enhancement of waters and related ecosystems for the benefit of present and future generations" (i.e. sustainable development). The PPWB Water Quality Agreement states that "in recognition of the need to use water quality objectives to maintain or enhance the quality of water and to avoid or resolve conflicts, the Board will apply water quality objectives to agreed upon interjurisdictional reaches."

PPWB water quality objectives encompass a broad range of uses that promote an integrated, ecosystem approach to


GOAL
sustainable development
IMPLEMENTATION
public attitude
policy
STANDARDS
SOE
EIA
resource management
MEASURES AND GOALS OF SUSTAINABILITY
WATER QUALITY OBJECTIVES
WATER QUALITY GUIDELINES
INCREASING INTEGRATION OF ENVIRONMENT/ ECONOMY DATA
stress-response
e.g. (eco)toxicology
ability of ecological base to support uses
resource valuation
DATA/INFORMATION
societal values
resource use
ecosystem characteristics
bio-physical data
economy
environment

adapted from Environment Canada (1990)

FIGURE 1. Building blocks for sustainable development.

water management. Uses considered in the formulation of objectives include domestic, aquatic life, wildlife, irrigation, industry, livestock, and recreation. Site-specific objectives are based on a consideration of all existing and potential uses, parameters of concern for protection of these uses, and identification of parameters that must be limited to protect uses. Ultimately, sitespecific objectives are developed which define maximum acceptable levels that should not be exceeded for each parameter, to protect the most sensitive use.

The PPWB example clearly illustrates the role and importance of water quality objectives in providing precise and scientifically defensible measures of sustainability that can be incorporated into the decision-making process. Importantly, they serve as the basis for a "regulatory" framework to protect and sustain water resources based on a broad, multi-functional perspective which integrates environmental and societal values.

Ecosystem Objectives for Lake Ontario

Over the past decade, our understanding of "water quality" has expanded to encompass the concept of aquatic "ecosystem health". Resources are part of a complex and interlinked system and long-term sustainability of our water resources and the many related uses and benefits depends on a holistic approach based on the protection of ecosystem integrity (Nelson and Eidsvik, 1990). Specific chemical objectives, though a critical component of a coherent water management framework, cannot be pursued out of context of the larger interactive system and water quality objectives must ultimately be set on the basis of long-term ecosystem needs (Rivers and Williams, 1989).

The development of ecosystem objectives for Lake Ontario is a prime example of the expanding nature and role of water quality objectives. The program involves several hierarchical components; a statement of goals based on societal values; ecosystem objectives for various ecosystem components required to meet the goals; and the development of quantitative indicators to measure progress toward each objective (Ecosystem Objectives Working Group, 1990).

Three goals for Lake Ontario have been identified as the basis for sustaining ecosystem integrity and uses on the long-term: 1) self-producing diverse biological communities: 2) levels of contaminants that do not limit the use of fish, wildlife and waters by humans or cause adverse health effects in plants and animals and: 3) recognition of society's impact on the ecosystem and responsible stewardship. To achieve these goals, five ecosystem objectives have been identified: 1) diverse, self-sustaining aquatic communities; 2) a diverse, self-sustaining wildlife community in the basin; 3) water, plants and animals free from contaminants at levels affecting human health or aesthetics; 4) sufficient quality and quantity of offshore, near-shore, wetland and upland habitat to support wild life and; 5) responsible stewardship.

The final step in the process is to develop the scientific measures (indicators) by which progress towards the broad ecosystem objectives can be evaluated.

Public Consultation and Education

"If the right environmental objectives are in place and the opportunity exists for society to respond in a rational fashion, then sustainable development with respect to water resources is attainable" (Rivers and Williams, 1989)

No matter how scientifically sound or laudable our goals for environmental sustainability may be, they will be ineffective if society does not participate fully in their development and delivery. Nelson and Eidsvik (1990) emphasize that an earlier shift towards an "eco-development" model was only partially successful because development and conservation were still basically envisioned as separate worlds. Sustainable development, by definition, serves to conceptually link these two worlds and the public must be involved if the process is to work.

Traditional water quality guidelines and objectives, by their nature, incorporate societal values and are communicated in terms that are readily understood and accepted by the public. In essence, water quality objectives may be seen as the "flesh on the bones" of the larger societal goals such as "fishable, swimmable water" (Hair, 1989). As water quality issues expand in scope and complexity to embrace ecosystem concepts, the public must play an increasingly important role in the development of water quality objectives to ensure that new and emerging goals reflect societal values and are understood and supported by the public.

Public participation is a cornerstone in the development of ecosystem objectives for Lake Ontario and has been incorporated fully into the process as mandated in the Great Lakes Water Quality Agreement. Part of the process was to identify the mechanisms by which the public input could be maximized. Several possible approaches were advocated including a public advisory committee, public workshops and hearings, public representation on work groups, public representation on committees, public notification with review and comments, correspondents and communication materials (Binational Objectives Development Committee, 1989). The consultation process is ongoing and a combination of these approaches has and will be used. For example, Great Lakes United has spear-headed public meetings throughout the Great Lakes region and the public has been invited to participate on work groups.

As water quality goals evolve to reflect a new understanding about the environment and our relationship to it, public education becomes an increasingly important part of the consultation process. Society must understand the stresses on the environment and the role and nature of objectives, if they are expected to participate effectively in the consultation process and to act in the ways necessary to achieve the objectives for a sustainable environment. Education and communication are an important component of the Great Lakes Program. If momentum towards the goal of sustainable development is to be maintained in light of increasing complexity in issues and actions, then both consultation and education must be considered as integral components of the process for development of water quality/ecosystem objectives.

Beyond the "End-of-thePipe": Water Quality Objectives in Environmental Management and Regulation

There is a growing recognition that the traditional "end-of-the-pipe" approach to water quality protection falls short of the goal to protect and sustain aquatic ecosystems (e.g. Rivers and Williams, 1989). Effluent discharge standards that specify contaminant concentrations do not take into account the total mass loadings to a water body that may ultimately exceed the assimilative capacity of the ecosystem. Neither can end-of-the-pipe standards account for the cumulative impact of several toxins from both point and non-point sources. There is a need for a more environmentally responsive approach to the regulation of industrial water pollution; one that incorporates water quality objectives for receiving waters.

Sprague (1990) describes a three-pronged regulation strategy for industrial water pollution that maximizes protection and maintenance of aquatic ecosystem integrity. His approach incorporates three tactics: 1) end-of-pipe (numerical limits for specified variables and toxicity); 2) site-specific (numerical limits for specified variables and sublethal toxicity at the edge of the mixing zone) and 3) ecological survey to assess effectiveness (changes in specific community variables). End-of-pipe limits provide a measure of control at the source but do not deal with important ecological questions and may not achieve environmental protection because characteristics of the receiving water body are not considered. Site-specific limits, which are analogous to the traditional guidelines/objectives values described earlier, overcome limitations of the first tactic and eliminate or prevent ecosystem impairment beyond the edge of the mixing zone. Sprague emphasizes that tactic two should be the strictest part of the enforcement package. Finally, ecological survey is recommended as a quality control check to ensure that the objectives applied at the edge of the mixing zone and at the end-of-pipe are effective in sustaining the aquatic community. This final tactic requires that we develop numerical values for "meaningful change" in communities (e.g. diversity, abundance) and whole organisms (physiological/biochemical) as indicators of ecosystem health. The need for such biological measures in addition to traditional chemical and physical measures of water quality, as part of a comprehensive water management strategy is becoming increasingly apparent as water quality goals broaden to encompass ecosystem protection and sustainability (e.g. EPA, 1990).

In Canada, pollution regulation and legislation is still largely entrenched in the first tactic (i.e. effluent standards). However, Canadian Water Quality Guidelines provide the basis for implementation of tactic two. The Great Lakes ecosystem work serves as a model for providing credible goals and measures for tactic three. If we are to deliver on the promise of sustainability, then more regulatory emphasis must be placed on tactics two and three (Sprague, 1990).

CONCLUSIONS

Water quality guidelines and objectives provide an important framework for sustainable development of water resources, defining clear and unequivocable goals and measures of sustainability that can be incorporated into the decision-making process. Though they are now extensively used across Canada, there is a need to incorporate water quality objectives more fully into the regulatory framework.

Emerging trends in the development of water quality objectives have followed the same general trajectory as our understanding of the environment and our relationship to it – shifting from an anthropocentric perspective ("drinkable, swimmable water") to an ecosystemic view that recognizes the need for ecosystem level goals and measures.

Ecotoxicology is a critical component of water quality guideline and objective development but the research base must be expanded at the community and ecosystem level. The development of biological measures of "water quality" and biological monitoring tools as a compliment to the traditional chemical objectives and measures is becoming an increasingly important component of a coherent framework for aquatic ecosystem protection.

There are important opportunities to build on the model provided by water quality guidelines and objectives in sustainable management of resources. Development of "environmental quality" guidelines that incorporate other media such as sediments and soils, and other resources such as wetlands and coastal waters, will provide the framework for a more comprehensive approach to sustainable development.

Finally, public participation is critical to ensuring that emerging ecosystem goals reflect long-term societal values and have the support and understanding of the public. It is a well accepted maxim that "where the people lead - the leaders will follow".

REFERENCES

Binational Objectives Development Committee 1989. Public Participation in the Parties Activities: Options and Recommendations. Unpublished draft report.

Canadian Council of Resource and Environment Ministers 1987a. Report of the National Task Force on Environment and Economy. September 24, 1987.

Canadian Council of Resource and Environment Ministers 1987b. Canadian Water Quality Guidelines. Prepared by the Task Force on Water Quality Guidelines.

Ecosystem Objectives Working Group 1990. Ecosystem Objectives for Lake Ontario. Unpublished draft report, March 1990.

Environment Canada 1990. Implementing Sustainable Development. Report of the Interdepartmental Workshop on Sustainable Development in Federal Natural Resource Departments. Mont Ste Marie, Quebec. June 1990.

Hair, J. D. (1989) Water Quality Standards for the 21st Century. In: Water Quality Standards for the 21st Century. Proceedings of a National Conference, March 1-3, 1989, Dallas, Texas. U.S. EPA, Office of Water, Washington D.C.

Hamner, R. (1989) Opening Address. In: Water Quality Standards for the 21st Century. Proceedings of a National Conference. March 1-3, 1989, Dallas, Texas. U.S. EPA, Office of Water, Washington D.C.

Nelson, J. G. and Eidsvik, H. E. 1990. Sustainable Development, Conservation Strategies and Heritage: Three Basic Tools for Influencing the Global Future. Alternatives 16(4) pp. 62-71.

O'Riordan, T. 1988. Politics of Sustainability. In: Sustainable Environmental Management: Principles and Practice. R. Kerry Turner ed., London, Belhaven Press.

Prairie Provinces Water Board 1990. Water Quality Procedures. Unpublished draft report. June 15, 1990.

Prairie Provinces Water Board 1989. PPWB Water Quality Indicators for the Churchill River at the Saskatchewan/Manitoba Boundary. Second Draft: January 1989.

Rivers, R. and Williams, D. 1989. The Role of Water Quality Objectives in Environment Economy Integration. In: Proceedings of a Workshop on the Development and Use of Water Quality Objectives. Task Force for Water Quality Guidelines, Canadian Council of Ministers of the Environment. Halifax, Sept. 19-21, 1989, Margaret C. Taylor (Ed.).

Shearman, R. 1990. The Meaning and Ethics of Sustainability. Environmental Management 14(1), pp. 1-8

Slater, R. W. 1988. The Challenge of Sustainable Development. Speaking Notes, Federal Real Property Conference, November 30, 1988.

Sprague, J. B. 1990. Environmentally Desirable Approaches for Regulating Effluent from Pulp Mills. 3rd IAWPRC Symposium on Forest Industry Wastewaters, Tampere, Finland, June 5-8, 1990.

SEDIMENT QUALITY GUIDELINES DEVELOPED FOR THE NATIONAL STATUS AND TRENDS PROGRAM

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The concentrations of selected potentially toxic chemicals in marine and estuarine sediments have been quantified annually by NOAA in the National Status and Trends (NS&T) Program since 1984. Sediments from about 200 sites nationwide have been sampled and analyzed for a variety of trace metals, petroleum hydrocarbons, and synthetic organic compounds. The chemical concentrations have been compared among sampling sites and among years at many of the sites. These data have been useful in characterizing the chemical conditions at sampling sites (NOAA, 1987, 1988) and in determining whether or not conditions are changing over time. In selected geographic areas measures of biological effects have been performed to accompany the chemical analyses and used to indicate the significance of the sediment contamination. However, biological measures of the effects or potential for effects of these mixtures of chemicals have not been determined at the majority of the sites.

Informal guidelines were developed for use the NS&T Program to put the data from new measurement into perspective. They were developed by employing a preponderance of evidence assembled from a variety of approaches and from data gathered in many geographic areas. These guidelines were used to rank and prioritize the NS&T Program sites with regard to the relative potential for contaminant-induced effects. These guidelines were not intended for use in regulatory decisions or any other similar applications.

The data that were assembled from 85 reports, the full explanation of the approach used and the guidelines for all NS&T Program analytes were reported by Long and Morgan (1990), available from the author. A brief synopsis of that report is presented in this extended summary.

OVERALL APPROACH

The approach involved assembling and reviewing currently available information in which estimates of the sediment concentrations of chemicals associated with adverse biological effects have been determined or could be derived; and determining the apparent ranges in concentrations of individual chemicals in which effects were often observed, based upon a preponderance of evidence. About 150 reports were reviewed. Of those, the data were used from about 85 reports in which either: (1) effects-based sediment quality values were reported, or (2)in which matching chemistry and biological effects data were listed, followed by an evaluation of the co-occurrence of chemical concentrations with measures of effects. These reports embraced controlled laboratory studies of effects of sediments spiked with individual chemicals (i.e., the spiked-sediment bioassay approach), calculations of unacceptable concentrations based upon theoretical equilibrium partitioning principles, and evaluations of data from field studies in which matching chemical and biological measures were performed on subsamples of sediments.

Chapman (1989) reviewed and compared the approaches currently being pursued to develop sediment quality values, but did not compare the concentrations resulting from those approaches. That report should be consulted for more information on each of the respective approaches. One approach not described by Chapman (1989) that was used extensively in the development of the NS&T Program guidelines was a co-occurrence approach. This method uses matching biological and chemical data collected in field studies. It involves calculation of statistics of central tendency (i.e., means, standard deviations, maxima, minima) in chemical concentrations associated with matching samples determined to have high, intermediate, and low indications of effects. Data were assembled from many studies conducted in many different geographic locations in North America and elsewhere. Both freshwater and marine data were used.

The data from the 85 reports were assembled and listed for each of the NS&T Program analytes according to the type of approach or type of test that was used, and, then, subjected to a screening step. If matching chemical and biological data from field studies showed no concordance, the data were not given further consideration. If no gradient (generally, less than a two-fold difference) in chemical concentrations was reported between samples that indicated adverse effects and those that did not indicate effects, the data for that particular chemical also were not given further consideration. If no definitive Apparent Effects Threshold (AET) concentration could be determined, the "greater-than" value reported was excluded from further consideration. The screening step was not performed to force consensus where none existed. The data that remained following this screening step were from studies in which effects were either predicted or observed in association with increasing concentrations of the respective analyte. Then, they were sorted in ascending order.

Next, two guidelines were determined from these remaining, sorted data for each chemical: an ER-L (Effects Range-Low), a concentration at the low end of the range in which effects had been observed; and an ER-M (Effects Range-Median), a concentration approximately midway in the range of reported values associated with biological effects. These two values were determined using a method similar to that used by Klapow and Lewis (1979) in establishing marine water quality standards for the State of California. For each chemical of interest, they assembled available data from spiked-water bioassays, examined the distribution of the reported LC50 values, and determined the lower 10- and 50-percentile concentrations among the ranges of values. In the present document, the ER-L values were concentrations equivalent to the lower 10 percentile of the screened, sorted data, and indicated the low end of the range of concentrations in which effects were observed or predicted. They were used in the document as the concentrations above which adverse effects may begin or are predicted among sensitive life stages and/or species or as determined in sublethal tests. The ER-M values for the chemicals were the concentrations equivalent to the 50 percentile point in the screened, sorted data. They were used in the document as the concentrations above which effects were frequently or always observed or predicted among most species.

In addition to the objectively determined ER-L and ER-M values, overall apparent effects thresholds were subjectively identified for some chemicals. These thresholds were the concentrations above which effects usually or always occurred in association with increasing concentrations of the chemical. They were determined independently of the ER-L and ED-M values by visually examining the sorted, ascending data tables. They are not to be confused with the AET values reported for Puget Sound, San Francisco Bay, and Mississippi Sound. They were identified as an aid in evaluating the accuracy of the ER-L and ER-M values and were not used in ranking the NS&T Program sites.

Data compilation and analysis was as inclusive as possible and no weighting was given to data derived from one approach or another. As Klapow and Lewis (1979) pointed out, the use of the inclusive approach and the calculation of percentiles of the data help eliminate the undue influence of a single (possibly outlier) data point upon the establishment of consensus ranges in concentrations associated with effects. In the present evaluation, the assumption was made that patterns established between effects and chemical concentrations would be more credible if based upon data from several sediment quality criteria than if based upon data from only one approach or experiment.

The relative degrees of confidence in the accuracy of the ER-L and ER-M values were subjectively judged for each analyte. Values for which we had relatively high confidence were those that were supported by clusters of data with similar concentrations, by data derived from more than one approach, by a data set that included more than results from the use of the co-occurrence approach, by data derived from multiple geographic areas, and for which the overall apparent effects threshold was similar to or within the range of the ER-L and ER-M values. Values for which we had relatively low confidence were those that were supported by data with either a small cluster or no cluster of similar concentrations, by data derived from only one approach and/or from one geographic area, results derived only from the co-occurrence approach, and for which the overall apparent effects threshold was dissimilar to or outside the range of the ER-L and ER-M values.

SUMMARY OF GUIDELINE CONCENTRATIONS

The ER-L and ER-M concentrations for each chemical or chemical group determined by Long and Morgan (1990) are listed in Table 1. These guidelines were used to evaluate the potential for toxic effects in sediments at the NS&T Program sites and to rank the sites.

The available data for some chemicals indicate agreements among the various approaches and the various data sets that were evaluated. For example, there is a relatively large amount of data available for cadmium generated from a variety of approaches and studies. The Puget Sound AET concentrations range from 5.1 ppm to 9.6 ppm; the 10-d LC50 concentrations from many spiked-sediment bioassays with amphipods range from 5.6 to 11.5 ppm; and significant toxicity to amphipods and reduced echinoderm abundance in Southern California sediments occurred in samples with mean cadmium concentrations of 5.3 and 6.2 ppm, respectively. Effects were not observed in sediments with cadmium concentrations of less than about 4 ppm. With some exceptions, biological effects were usually observed in association with cadmium concentrations of 5 ppm or greater. The preponderance of evidence from these data suggest that effects are likely or expected as cadmium concentrations in sediments reach about 5 ppm. Also, the effect of adding or deleting data upon the ER-L and ER-M values for cadmium would likely be relatively small.

For some other chemicals, there was less agreement among the data from various approaches and the degree of confidence in the accuracy of the resulting ER-L and ER-M values was relatively low. For example, the Puget Sound AET concentrations for chromium are 260 and 270 ppm, whereas effects were observed elsewhere in association with mean concentrations as low as 61 ppm and as high as 1646 ppm. Many of the biological measures of effects were not in concordance with chromium concentrations, suggesting that chromium had a minimal role or no role in causation. In another example, the Screening Level Concentrations for total PCBs range from 2.9 ppb to 42.6 ppb based upon a relatively large amount of data; whereas, the Puget Sound AET concentrations range from 130 ppb to 3100 ppb, the San Francisco Bay AET range from 54 to 260 ppb, the chronic marine threshold predicted by Equilibrium Partitioning methods is 280 ppb, and the LC50 from a spiked sediment bioassay performed with amphipods is 10800 ppb. The effect of adding or deleting data upon the ER-L or ER-M values could be significant for some of the chemicals for which there is little consistency or clustering in the data. Obviously, for many chemicals there is yet much to be learned as regards the chemical concentrations in sediments that cause biological effects.

Overall, the degree of confidence in the accuracy of the ER-L and ER-M values should be considered as moderate for the metals group and PCBs and low for the pesticide and PAH groups. Much more data are needed to support or refute the ER-L and ER-M values for all groups and for individual analytes within the groups. Since no data were available for tropical areas, for most of the southeastern USA, or for subarctic areas, the applicability of these guidelines to those areas is uncertain.

Also included in Table 1 is a summary of the subjectively determined, overall apparent effects thresholds for each chemical; the concentrations equal to and above which biological effects were usually or always observed. The ER-L and ER-M values were established objectively with *a priori* selection criteria, i.e., the lower 10 percentiles and 50 percentiles of the available data. They were not established following review and evaluation of the data for each chemical. However, following a review of the available data for each chemical, apparent effects thresholds were often observed and noted. These thresholds were established with a subjective approach. Therefore, they were identified and listed as evidence to support the strength of the ER-L and ER-M values and as hypotheses to be evaluated with additional data. They were not used to rank the NS&T Program sites. For several chemical analytes (i.e., chromium, total DDT, dieldrin), there was no apparent effects threshold. No effects were observed in some studies where high concentrations of these chemicals occurred. For nickel and many of the pesticides and aromatic hydrocarbons, there were insufficient data to determine a threshold, noted as not sufficient data (NSD) in Table 1. For many of the analytes, e.g., mercury, there were inconsistent data at concentrations above the apparent effects thresholds, i.e., data from some studies indicated no effects at relatively high concentrations of the analyte. There was a very distinct threshold for some chemicals, i.e., arsenic, cadmium, copper, lead, mercury, zinc, PCBs, anthracene, and some other aromatic hydrocarbons. The apparent effects thresholds for most of the trace metals, PCBs, DDT, and some of the aromatic hydrocarbons were very similar to the respective ER-M values or within the ER-L/ER-M range. However, the apparent threshold was outside the ER-L/ER-M range for antimony and lead.

SUGGESTED USES OF THE GUIDELINES

Since the report by Long and Morgan (1990) was published, the ERL and ERM guidelines have been used by others in applications other than the evaluation of the NS&T Program data. Some questions have arisen as a result of others using the guidelines. Clarification of the suggested uses of the data reported by Long and Morgan (1990) are, perhaps, in order.

First, instead of using the ERL and ERM values alone, it is suggested that investigators also compare their new data with the ascending, sorted data presented in Appendix B of the report. Determine how the new data compare with observations or predictions made by others. Determine if the concentrations in the new data equal or exceed the concentrations previously associated with toxic effects or not, and note what types of effects were observed and the approach that was used. For example, assume that the chemical analysis of a sample indicated that it contained 4 ppm fluoranthene. Examination of the data in Appendix B-25 of Long and Morgan (1990) indicates that this concentration exceeds the ERM value (3.6 ppm), the overall apparent effects threshold (1 ppm), an LC50 from a spiked sediment bioassay, the mean concentration associated with effects in Commencement Bay and San Francisco Bay, AET concentrations determined with a variety of different biological tests, marine screening level concentrations, and the toxic concentrations predicted with the equilibrium partitioning approach. With this weight of evidence, one could assume that the sample with 4 ppm likely would present a toxicological problem.

Second, the investigator should examine the data tables in the report to determine how applicable the data may be to the evaluation of his/her new measurements. That is, new data collected from the southeastern portion of the USA, for example, may not be comparable with those assembled in the tables. Also, the investigator may have low confidence in certain approaches or particular data sets that were used by Long and Morgan (1990), and may elect to ignore or reject some of those data.

Third, it should not be assumed that the data used in the document necessarily reflect cause/effect relationships. The approaches that were reported often reflected merely the association between the concentrations of chemicals and the observations of toxic effects in the same samples. No cause/effect relationship was necessarily implied.

Long and Morgan (1990) accepted and used data from many different studies that undoubtedly relied upon different analytical methods. Sediment bioassays were performed with many different species and included different toxicological end-points. Extraction and analytical methods for chemical analyses differed among the studies. The models used to predict equilibrium-partitioning constants improved and changed iteratively as new information was acquired. Some biogeographic areas are not represented by any data. Freshwater and marine data were merged and treated as if they were comparable, whereas some toxicants may not be equivalent in bioavailability in the two regions. No attempt was made to normalize the chemical data to total organic carbon, grain size, acid volatile sulfides or any other parameters. Therefore, caution is advised in using the ERL and ERM guidelines unequivocally as absolutes. They should be used as informal guidelines and should be used along with the evidence listed in the ascending, sorted tables.

Finally, it should be understood that there is a need for additional information for all of the analytes. There seems to be very little consensus among the data for some chemicals. The reasons behind this lack of agreement should be explored and determined. On the other hand, relatively good agreement is apparent for some chemicals - additional data should be acquired to further test (and verify) the effects range despite the occurrence of a consensus.

REFERENCES

Chapman, P. M. 1989. Current approaches to developing sediment quality criteria. *Environmental Toxicology and Chemistry* 8: 589-599.

Klapow, L. A. and R. H. Lewis. 1979. Analysis of toxicity data for California marine water quality standards. *Journal Water Pollution Control Federation 51*(8): 2051-2070.

Long, E.R. and L.C. Morgan. 1990. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. NOAA Tech. Memo. NOS OMA 52. Seattle, WA. National Oceanic and Atmospheric Administration. 175 pp. + appendices.

National Oceanic and Atmospheric Administration. 1987. National Status & Trends Program for Marine Environmental Quality Progress report and preliminary assessment of findings of the Benthic Surveillance Project-1984. Rockville, MD: Office of Oceanography and Marine Assessment, National Oceanic and Atmospheric Administration. 81 pp.

National Oceanic and Atmospheric Administration. 1988. Progress Report. A summary of selected data on chemical contaminants in sediments collected during 1984, 1985, 1986, and 1987. NOAA Technical Memorandum NOS OMA 44. Rockville, MD. National Oceanic and Atmospheric Administration. 15 pp.

DESIGN OF THE GULF OF MAINE MARINE ENVIRONMENTAL QUALITY MONITORING PROGRAM

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Under the direction of the five jurisdiction Gulf of Maine Council on the Marine Environment, a monitoring program was designed to meet three goals: provide information on the status, trends and sources of human health risks; provide information on the status, trends and sources of risks to ecosystem integrity; and provide this information in an appropriate and timely manner to managers. The program is designed to integrate and enhance numerous existing municipal, state, provincial and federal programs, including the NOAA Status and Trends Program, the FDA National Shellfish Sanitation Program, and the CWS Seabirds Contaminants Program in addition to existing effluent permitting programs. Information will be integrated through use of a personal computer-based link designed to use the existing data base management system in those agencies. Information interpretation will be completed by the Council's Working Group. Given that one of the options is accepted by the Council, a pilot project will be implemented as early as 1991.

SUMMARY

The Gulf of Maine is no longer pristine. Tons of raw and partially treated sewage are discharged into the Gulf each day. Industrial discharges and urban and agricultural runoff introduce toxic contaminants and pathogens to marine and estuarine waters on a chronic, and at times, acute, basis. Increased fishing effort has reduced fish stocks to all time lows. Coastal development has encroached on environmentally significant marine wetlands. Accidental spills of oils and other toxic material place additional stresses upon the Gulf environment.

To understand and manage the impact of such stresses on the health of the Gulf ecosystem requires accurate understanding of the nature, scale, and impact of environmental perturbations in the Gulf. As a step toward generating the requisite information, the Gulf of Maine Council on the Marine Environment has established a tightly focused and pragmatic environmental quality monitoring plan for the Gulf of Maine.

The Gulf of Maine council on the Marine Environment was established by the Governors of Maine, Massachusetts and New Hampshire, and the Premiers of Nova Scotia and New Brunswick to improve the environmental management of the Gulf of Maine.

The Council has identified assessment of the health of the Gulf as of pressing importance. The council initiated development of a monitoring plan as a first step toward improving environmental management of the Gulf, envisioning a program that will allow evaluation of environmental quality of the Gulf while improving the effectiveness of prevention and remediation efforts.

The monitoring plan is based on a mission statement provided by the Council:

> It is the mission of the Gulf of Maine marine environmental quality monitoring program to provide environmental and resource managers with information to support sustainable use of the Gulf, and allow assessment and management of risk to public and environmental health from current and potential threats.

The Council charged the Gulf of Maine Working Group, the management component of the council, and its monitoring sub committee with identifying the environmental quality issues of greatest importance to the Gulf States and Provinces and with developing a monitoring plan to address these issues.

As part of this process, at the December 1989 *Conference on Sustaining the Common Heritage of the Gulf of Maine*, working sessions were devoted to monitoring and a workshop was held in Halifax in early June, 1990 to review a draft report on the proposed monitoring program. Scientists, environmental managers and policy-makers from throughout the Gulf region worked together to develop consensus on goals and objectives and to begin the process of identifying priorities and selecting appropriate monitoring methodologies. The current plan reflects the results of these consultations.

Three monitoring goals were established to meet the Program's Mission:

- To provide information on the status, trends and sources of risks to the marine environment in the Gulf of Maine;

- To provide information on status, trends, and sources of marine-based human health risks in the Gulf of Maine; and

- To provide appropriate and timely information to environmental and resource managers that will allow both efficient and effective management action and evaluation of such action.

Several objectives were developed to meet these goals and are listed in the planning document. To narrow the scope of the monitoring plan the objectives were ranked in order of importance; the plan was developed to address only the top three objectives. The remaining objectives will be addressed as the program is implemented and resources are available. In order to rank the priority of the monitoring goals and objectives, a survey was distributed at the December 1989 Conference and mailed to over 150 scientists, environmental managers, policy makers, and others in the Gulf. The rankings were further discussed at the workshop in Halifax. The three objectives with the highest priority were:

1. To assess the status and trends in the marine environment by monitoring appropriate indicators, especially those that will allow early identification of change in environmental quality;

2. To assess the existing levels, the trends, sources, and economic impacts of acute and chronic risks to human health from toxic compounds transmitted through marine foods and water contact; and

3. In cases where environmental degradation is suspected, to identify the probable causes, especially as they reflect anthropogenic impacts and cumulative effects.

The participants of the workshop identified an ancillary priority that is not related specifically to monitoring, but should be an important part of any environmental management strategy for the Gulf: all existing environmental data on the Gulf must be organized, assessed for quality, and made accessible to a wide range of users.

The monitoring plan described outlines only the monitoring methods needed to meet the three highest priority monitoring objectives. It then identifies ongoing monitoring programs in the Gulf that are addressing these objectives.

The Monitoring Subcommittee of the Working Group intends to establish the broader goals and objectives of a Gulf -wide environmental quality monitoring program as the first step in the development of such a program. Further implementation of a monitoring program will require establishing specific monitoring methodologies, setting acceptable levels of precision and accuracy, and developing detailed sampling designs which specify the number of samples to be collected, the exact locations, and the laboratory procedures to be used for analyzing the samples. These are details best left to practitioners in the field.

Ad hoc committees will be formed to identify specific implementable monitoring methodologies for the priority objectives identifies in the plan. In addition, the plan will be reviewed at a major scientific conference on the Gulf of Maine and at this Symposium.

The monitoring methods presented here are developed from a review of methods currently used in other programs and from a list of monitoring questions that were developed to address each objective. The questions are categorized in terms of six monitoring parameters which include:

1. The variable to be monitored;

2. The sampling medium in which the variable is measured (i.e. soft bottom, hard bottom tissue etc.);

3. The geographical scale/location where sampling should take place;

4. The frequency with which the variable should be monitored;

5. The field methods to be used to monitor the variable; and

6. The type of data analyses needed to provide the information to answer the monitoring question.

The plan includes identification of the proposed areas where monitoring should take place and the estimated cost of monitoring other appropriate variables. In all cases cost estimates are based on the assumption that the private sector will do the work. It is estimated that monitoring a broad range of indicators to meet the first objective (assessing the marine environment) will cost of $3,000,000 US annually. This assumes that existing monitoring programs can be modified as needed to collect the appropriate data in their current study areas. The estimated cost for collecting the information needed to meet the objective with the second highest priority (assessing human health risk) cost estimate, however, is based on only monitoring the risks of mercury and PCB's, the two toxic compounds for which standards in foods have been developed. There is a major need to fund additional research to understand the human health risks from other toxic compounds. The costs for meeting the objective with the third highest priority, that of identifying causes, cannot be estimated at present because the area and scale of environmental changes have not yet been identified.

The plan also outlines four additional aspects of a monitoring program:

- The procedures to facilitate the transfer of information between the scientists analyzing the monitoring data and the environmental managers who will be using the information to develop management actions;

- Guidelines for developing a database for storing the information collected by the monitoring program; and

- An implementation plan incorporating a pilot program utilizing the "Mussel-watch" concept.

As a strategy for implementation, the plan will build on monitoring activities currently underway in the Gulf. For example, it is anticipated that the Status and Trends Program of the U. S. National 0ceanic and Atmospheric Administration will be expanded to answer questions about the health of the larger Gulf ecosystem. Gaps in existing programs will be identified and new programs designed. In addition, the plan will integrate local problems, such as shellfish closures, that occur throughout the Gulf region. Data collected from coastal embayments on toxic contamination, nutrient enrichment, and shellfish and beach closures will be augmented by similar data collected in other industrialized embayments along the Gulf shore. It is our hope that this collective approach will yield better solutions to problems encountered or anticipated in such areas.

The success of this endeavour will depend on:

- The cooperation of State, Provinces and federal agencies in adapting existing monitoring programs to serve the objectives of the Gulf program as well as their own objectives;

- Funding for new monitoring to fill gaps identified in existing monitoring activities.

- Regional coordination to provide guidance for the development and implementation of the program;

- A database management system that will allow information generated by the monitoring program to be readily available to environmental managers throughout the region; and

- Links to a geographic information system such as Environment Canada's FMG project.

The Monitoring Subcommittee invites your comments on this monitoring plan. Further development of this plan requires the informed participation of monitoring professionals, other scientists, environmental managers, and policy-makers. Please forward your comments to the Monitoring Subcommittee, c/o Maine State Planning Office, Station 38, Augusta, Maine 04333, so that they may be incorporated in further iterations of the plan.

LESSONS FROM U.S. NATIONAL AND REGIONAL MONITORING PROGRAMS

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Long-term inconsistency and poor availability of data and supporting information, are two major problems inhibiting successful assessment of long-term pollution trends in coastal waters of the United States. Large-scale geographical and long-term trends of contaminant concentrations in fish, shellfish, and sediments were developed from recent and historical data collected from a dozen national and several hundred regional and state monitoring programs conducted aperiodically over the past 25 to 30 years. National monitoring programs, involving up to 200 sites, were underway between 1965 and 1972, 1976-78, and 1985-present under various agency sponsorship. State and regional monitoring programs complimented federal programs in some areas (Maryland, Texas, California) but other areas were neglected by both national and state agencies. Reconstructions revealed 100-fold declines of DDT and other pesticides on a national basis during the past 20 years, while PCB concentrations declined only near well-known major point sources. Regional analyses revealed no substantial changes in concentrations of most metals in biota, despite marked declines in inputs and sediment concentrations supporting the idea that metals have not been important contaminants of the sea coast.

Once a major monitoring program begins, all effort should be made to continue it and to resist change even in the face of changing technology. To date, there remains no national or regional commitment to establishing, on a continuing basis, data management systems that accommodate historical as well as current data and supporting information. It is not necessary that all such data be digitized-availability alone is half the battle!

POLYCHAETES: KEY TAXA IN MARINE ENVIRONMENTAL QUALITY MONITORING

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Funding for this project provided by: P.G. Wells, Senior Advisor, Marine Environmental Quality, Conservation and Protection, Department of the Environment, Dartmouth, N.S.

One primary objective of biomonitoring is to assess the impact of man-made changes on the biosphere. From a biological perspective this can be achieved in two ways: 1) by examining the body burdens and the effects toxic materials have on representative organisms, 2) by examining the biota affected by contaminants. Polychaetes serve both of these functions, in the first case, as bioassay organisms and monitors for contaminants; in the second, as pollution indicators at the species, population or community level.

This study reviews recent literature pertaining to polychaetes useful in marine environmental quality monitoring, it analyses their effectiveness in this role, and it examines the potentially greater use of polychaetes as biomonitors for regulatory and marine environmental quality monitoring purposes.

ECOTOXICOLOGICAL TESTING

The Bioassay

Polychaetes fulfil the requirements of bioassay organisms. They are indigenous, ecologically significant and easily collected. Some species are dependant upon the quality of interstitial water and sediment particles. Results obtained using polychaetes as bioassay organisms are compatible with other test species (Chapman et al. 1985, McIlroy and Means 1988, Becker et al. 1989, and Jop 1989). Polychaetes have given responses to a) heavy metals; b) organic compounds; and c) radiation; by changes in: 1) fecundity (Carr et al. 1989), 2) reproduction (Reish 1977, Jop 1989, Reish 1980, and Oshida et al. 1981); 3) growth (Røed 1981); 4) survival (Swartz et al. 1979, Pesch et al. 1986).

Indicators of Bioaccumulation

As monitoring organisms, polychaetes have been shown to accumulate deleterious materials such as a) heavy metals (Reish 1984, Bryan and Gibbs 1987), b) organic compounds such as PCB's, PAH's, HCBP's (McLeese et al. 1987, and McElroy 1988) and c) organic metal complexes (e.g. organotin) (Langston et al. 1987), within their bodies, in concentrations proportional to the concentrations found in the environment. This makes them good indicators for the presence and bioaccumulation potential of these materials.

BENTHIC FAUNAL ANALYSIS

Indicators at the Species, Population and Community level.

Polychaetes are sensitive to organic enrichment, toxic material and heavy metal deposition, hence it follows that the presence or absence of specific polychaetes in contaminated areas are an excellent indication of marine environmental quality. Species of the family *Capitellidae* and *Spionidae* are widely accepted as pollution indicators; species of other families, e.g. *Nereidae* and *Nephtyidae*, are accepted as indicators of early successional phases in recovery (Pearson and Rosenberg 1978).

At the population level, a variation from the expected distribution of year classes can indicate a stress on a population, or enormous numbers of one species to the exclusion of all others can indicate deleterious environmental conditions.

At the community level, numerous indices which show changes in diversity, abundance, biomass, dominance and numerical distribution have been used to show quantitatively that the community is under stress. Polychaetes are usually the most abundant organisms and account for the greatest number of species in many deleterious environmental conditions and therefore are a significant element in these analyses. An Annelid Pollution Index has been devised and used to assess pollution at a site affected by municipal sewage (Bellan 1980, Bellan et al. 1988).

An excellent use of analysis of the benthic community is to show a gradient in the community composition corresponding to a gradient in physico-chemical characteristics. This has been done from a point source for sewage (Holte and Gulliksen 1987), for hydrocarbon concentrations near a drilling rig (Addy et al. 1984) and to determine the extent of enrichment away from a fish farm (Weston 1990).

Polychaetes in Multi-parameter programs

Polychaetes both as laboratory test animals and field monitoring organisms can be used alone or in combination with other monitoring processes such as the Benthic Triad (community structure, sediment bioassay and sediment chemistry).

Recommendations for Canadian Status and Trends Monitoring Programs

Polychaetes should be:

1) Used for bioassay toxicity testing;
2) Used as monitoring organisms for contaminants;
3) Used as indicator organisms at the species, population and community level, alone or as part of a multi-parameter program, once base line information on their status in Canadian waters is established.
4) The study of polychaetes should be an integral part of industrial effluent monitoring programs to be conducted in Canadian coastal waters.

REFERENCES

Addy, J. M., J. P. Hartley and P. J. C Tibbets. 1984. Ecological effects of low toxicity oil-based mud drilling in the Beatrice oilfield. Mar. Poll. Bull. 15(12): 429-436.

Becker, D. S., T. C. Ginn and G. R. Bilyard. 1989. Comparisons between sediment bioassays and alterations of Benthic macroinvertebrate assemblages as measures of sediment toxicity. in. Oceans '89, the Global Ocean, Vol.2. Ocean Pollution pp. 461-466.

Bellan, G. 1980. Relationship of pollution to rocky substratum polychaetes on the French Mediterranean coast. Mar. Poll. Bull. 11: 318-321.

Bellan, G. G. Desrosiers and A. Willsie. 1988. Use of an Annelid Pollution Index for monitoring a moderately polluted littoral zone. Mar. Poll. Bull. 19(12): 662-665.

Bryan, G. W. and P. E. Gibbs. 1987. Polychaetes as indicators of heavy-metal availability in marine deposits. in. Oceanic Processes in Marine Pollution Vol 1, Chapter 4 pp. 37 -49.

Carr, R. S., J. W. Williams and C. T. B. Fragata. 1989. Development and evaluation of a novel marine sediment pore water toxicity test with the polychaete *Dinophilus gyrociliatus*, Environmental toxicology and chemistry 8: 533-543.

Chapman, P. M., R. N. Dexter, R. M. Kocan and E. R. Long. 1985. An overview of biological effects testing in Puget Sound, Washington: Methods, Results and Implications. Aquatic Toxicology and Hazard Assessment: Seventh Symposium, ASTM STP 845, R. D. Cardwell, R. Purdy and R. C. Bahner, Eds. pp. 344-363.

Holte, B. and B. Gullikson. 1987. Benthic communities and their physical environment in relation to urban pollution from the city of Tromso, Norway. Sarsia 72: 133-141.

Jop, K. M. 1989. Acute and rapid-chronic toxicity of hexavalent chromium to five marine species. Aquatic Toxicology and Hazard Assessment: 12 th volume. ASTM STP 1027, U. M. Cowill and L. R. Williams eds. pp. 251-260.

Langston, W. J., G. R. Burt and M. J. A. F. Zhou. 1987. Tin and organotin in water, sediments, and benthic organisms of Poole Harbour. Mar. Poll. Bull. 18(12): 634-639

Long. E. R. and P. M. Chapman. 1985. A sediment quality Triad: Measures of sediment contamination, toxicity and infaunal community composition in Puget Sound. Mar. Poll. Bull. 16(10): 405-415.

McElroy, E. 1988. Trophic transfer of PAH and metabolites (fish, worm). Crisp Data Base, National Institutes Of Health.

McElroy, A. E. and J. C. Means. 1988. Factors affecting the bioavailability of hexachlorobiphenyls to benthic organisms. Aquatic Toxicology and Hazard Assessment: 10 th Volume, ASTM STP 971, W.J. Adams, G. A. Chapman and W. G. Landis, Eds. pp. 149-158.

McLeese, D. W., L. E. Burridge, J. M. Capuzzo and D. R. Kester. 1987. Comparative accumulation of PAHs in four marine invertebrates. in Oceanic Processes In Marine Pollution. Vol. 1: Biological Processes and Wastes in the Ocean. pp. 109-118

Oshida, P. S., L. S. Ward, and A. J. Mearns. 1981. Effects of hexavalent and trivalent chromium on the reproduction of *Neanthes arenaceodentata* (Polychaeta). Mar. Environ. Res. 7: 167-174

Pearson, T. H. and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment of the marine environment. Oceanogr. Mar. Biol. Ann. Rev. 16: 229-311.

Pesch, C. E., P. S. Schauer and M. A. Balboni. 1986. Effect of diet on copper toxicity to *Neanthes arenaceodentata* (Annelida: Polychaeta). Aquatic Toxicology and Environmental Fate: Ninth Volume. ASTM STP 921, T. M. Poston and R. Purdy, Eds. pp. 369-383.

Reish, D. J. 1977. Effects of chromium on the life history of *Capitella capitata*. In: Physiological Responses of Marine Biota to Pollutants. Vernberg, F. J., A. Calabrese, F. P. Thurberg and W. B. Vernberg (Eds). Academic Press New York. pp. 199-207.

Reish, D. J. 1980. The Effect of Different Pollutants on Ecologically Important Polychaete Worms. U.S. Environmental Agency, Environmental Research Laboratory, Narragansett, R.I. EPA 600/3-80-053. 138 pp.

Reish, D. J. 1984. Marine Ecotoxicological Tests With Polychaetous Annelids. In Ecotoxicological Testing For The Marine Environment. G. Persoone, E. Jaspers and C. Claus (Eds.) State Univer. Ghent and Inst. Mart. Scient. Res. Bredene, Belgium Vol.1. 798 pp.

Roed, K. H. 1980. Effects of salinity, and cadmium on reproduction and growth during three successive generations of *Ophryotrocha labronica* (Polychaeta). Helgolander Meeresunters. 33: 47-58.

Swartz, R. C. 1987. Toxicological methods for determining the effects of contaminated sediment on marine organisms. In: Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems. K. L. Dickson, A. W. Maki and W. A. Brungs (Eds.) pp. 183-198.

Weston, D. P. 1990. Quantitative examination of macrobenthic community changes along an organic enrichment gradient. Mar. Ecol. Prog. Ser. 61: 233-244.

APPENDICES

APPENDIX 1

SUMMARY: NOTES OF WORKSHOP ON MARINE ENVIRONMENTAL QUALITY MONITORING

Chaired by Michael Waldichuk and Matrin Pomeroy

7 November 1990, Aquatic Toxicity Workshop, Vancouver, B.C.

Michael Waldichuk (chair) referred to several issues raised in *Monitoring Troubled Waters: The Role of Environmental Monitoring* (National Research Council, USA, National Academy Press):

Failure to produce desired effect in marine environmental monitoring is attributable to three factors:

A. poor design of the monitoring program and inappropriate application of monitoring technology
B. lack of presentation of monitoring information in a form that is useful for evaluating specific control strategies
C. limits on scientific knowledge and predictive capabilities.

Sound monitoring design and implementation depend on the following:

1. The goals and objectives of the monitoring program must be clearly articulated in terms that pose questions that provide the basis for scientific investigation.
2. Attention must be paid not only to gathering of the monitoring data but also to their management, synthesis, interpretation and analysis.
3. Quality assurance procedures must be rigorously followed.
4. Supportive research should be provided to answer questions about environmental research and human impacts raised by well designed monitoring programs.
5. Adequate resources are required for detailed analysis of monitoring data.
6. Programs should be sufficiently flexible so that changes can be made if new information suggests that these are needed.
7. Monitoring information should be made available to all interested parties in a form that is useful to them.

Papers were presented on *Design of the Gulf of Maine Environmental Quality Monitoring Program* (Barchard and Johnson Hayden); *Monitoring the coastal and estuarine health of the United States using bi-valve molluscs as sentinel organisms* (Lauenstein); *A Study of Benthic Contaminants in Vancouver Harbour B.C. to Assess the Environmental Quality* (Goyette); and *Bacteriological Monitoring of Shellfish in B.C. Waters* (Kay).

Discussion was focused on three questions:

1 **What is an appropriate program focus for a marine environmental monitoring program?**
  - management objectives as mandated, for example, by statute?
  - environmental parameters defining environmental quality, trends, etc?

2 **Where do existing programs fit in?**
  - can they be used or interpreted for use as monitoring programs?
  - can existing data management systems be borrowed from programs which have a different focus?

3 **What should be included in a monitoring program? What are useful indicators?**
  - ambient environmental conditions?
  - site specific criteria?
  - State of the Environment data?

4 Program Focus

- different client groups may have different needs
  - **managers** need information to evaluate programs
  - **the public** wants state of the environment information: "how bad is it, is it getting better or worse, what's being done"
  - **environmental scientists** need data to understand how the environment works
- defining the program focus will help define what is to be included and how to make use of existing programs
- some problems:
  - managers' changing priorities can lead to inconsistency and discontinuity in data
  - "the public" has many different aspects, from environmental to business pressure groups, and not all of them are in agreement
  - meeting the official "mandate" of the managers sometimes fails to meet what the public is asking for
  - not all the questions that are asked are clearly answerable – it's a waste of money to pursue questions that are too vague
- some approaches:
  - ask the community what it wants (as in Halifax harbour study, Gulf of Maine) in order to establish goals
  - an agreement between management and other groups (the public, scientists, national and international bodies) would help to define a program's focus and goals, and give it long term continuity
  - a steering committee composed of management and representatives of the public (and scientists?) running the program after it is established would help give continuity, consistency and direction

5 Existing programs

- not many meet the needs of an environmental monitoring program, with possible exceptions such as the seabird monitoring and "mussel watch" programs
- most existing programs have been monitoring for specific parameters such as fish production or presence of metals in fish, but these may not lead to useful conclusions about the environment
- existing raw data sets can be analyzed and interpreted to see if they provide useful and relevant information, or information that can be adapted, for environmental monitoring
- quality assurance/quality control are important factors but are difficult to define for existing data
- it is useful to develop a data base prior to information collection to ensure that data can be effectively handled and compared, rather than trying to fit varying existing data sets into a newly created data base
- by formalizing protocols and adding missing elements to existing programs, it may be possible to create data that can more easily be compared to each other

6 What tests?

- this depends on what goals are defined as program focus
- there is no universal monitor – a "suite" of indicators or series of regionally specific indicators may be required
- monitoring for biological effects may be the most useful approach, as it is difficult to interpret environmental effects from narrower, more specific tests
- no agreement on what biological tests, however, it is important to monitor habitat and external factors as well, in order to understand changes in results of monitoring.

APPENDIX 2

MUSSEL WATCH PANEL DISCUSSION

Hotel Vancouver, November 7, 1990 Chaired by: Bruce Kay and Elaine McKnight

SUMMARY OF DISCUSSIONS

Background

In early 1990 the Shellfish Water Quality Protection Program of Environment Canada contracted to Seakem Oceanography Ltd. to prepare a review document of Mussel Watch Programs in North America and elsewhere in the world. The purpose of the contract was to provide an evaluation of the benefits and problems associated with Mussel Watch programs, and provide a basis for further discussions within Environment Canada and other federal agencies (such as the Department of Fisheries and Oceans) as to the potential for beginning a Mussel Watch program in Canada.

Using the draft report developed from this contract ("National Mussel Watch Program. Phase I. Overview of Existing Mussel Watch Programs") as background material, a Panel Discussion was conducted in conjunction with the 17th Annual Aquatic Toxicity Workshop in Vancouver to evaluate specific objectives for a Mussel Watch program and discuss a number of technical issues raised by the Seakem report. These discussion points are presented in Appendix I. The Panel was comprised of a number of experts in Mussel Watch studies as well as program managers for the Shellfish Program and other marine environmental quality programs. A list of participants is given in Appendix II. This report summarises these discussions.

Mussel Watch Objectives

The objective presented to the Panel for discussion was to develop a program to measure contaminant concentrations in bivalves in approved shellfish growing areas as part of the requirements of the Canadian Shellfish Sanitation Program. The intent was to develop a baseline range of reference levels, establish a national data bank, identify potential "hot spots" and establish trends in contaminant data. Expansion of the monitoring network to include other areas as part of a more complete status and trends program would be considered if the pilot growing area program was successful.

The Panel discussed whether the objectives as presented represented a human health protection monitoring strategy or an environmental (status and trends) monitoring strategy. It was generally agreed that both purposes would be served but primarily it would provide environmental data which health officials and environmental managers could use. It was pointed out that Health and Welfare Canada does not have sufficient environmental baseline data to assist in identifying human health threats.

The panel generally agreed that the objectives were suitable to a pilot project for approved growing areas and recommended the addition of other reference sites which would be considered more integrative in nature. Such sites could include areas presently closed to direct harvesting but suitable for relay or depuration harvesting. It was also recommended that some of these sites parallel the Contaminants in Seabirds monitoring sites established by the Canadian Wildlife Service.

Technical Considerations

The Panel was presented with a number of technical questions on the following subjects:

1. Sampling design
2. Target species
3. Contaminants of Concern
4. Protocols (Analytical)
5. Statistical Techniques
6. QA/QC
7. Costing

The discussions concentrated on site selection, target species, use of caged mussels, other physiological measures and contaminants of concern. After considerable discussion the Panel came to the following general conclusions:

1. The present NOAA National Status and Trends Program Mussel Watch has dealt with most of the technical issues in the design of its program and should be chosen as the model for the any Mussel Watch program contemplated for shellfish growing areas (i.e. don't re-invent the wheel).

2. Identification of contaminants of concern should consider the following:

- U.S. Mussel Watch contaminants list is a good starting point
- specific regional chemical use patterns (eg. pesticides) must be identified
- contaminants measured in the CWS Contaminants in Seabirds program should also be measured in a Mussel Watch
- an initial broad screening of contaminants should be undertaken the first year and priority pollutants identified for continuing measurement

Conclusions and Recommendations

1. The Panel concluded there was sufficient merit in proceeding with a program to measure contaminant levels in bivalve molluscs and sediments from shellfish growing areas and other reference sites.

2. It was agreed that a Steering Committee be struck to develop a protocol document for a Mussel Watch Pilot Project which would address: site selection; target species; sampling/collection requirements; contaminants of concern; analytical procedures; QA/QC; statistical requirements and constraints; physiological considerations and other measurements. Several of the Panel members agreed to participate in this Steering Committee. The protocol document would use the NOAA Status and Trends Program Mussel Watch as the reference program.

3. A Terms of Reference be developed for the Steering Committee within 2-3 weeks with a draft protocol document to be available by the end of January 1991 for review by Shellfish Program managers.

4. A final document detailing protocols, integration with other programs and initial and annual costs be completed by the end of March 1991.

APPENDIX I

ENVIRONMENT CANADA'S OBJECTIVES FOR A MUSSEL WATCH PROGRAM

1. Monitoring of the concentration of contaminants in bivalves in shellfish growing areas should strengthen National Shellfish Sanitation Program and Canadian Shellfish Sanitation Program requirements by providing DOE with data on the concentration of contaminants in bivalves to meet obligations in the classification of these growing areas.

2. Within shellfish growing areas

a) A "baseline" or range of reference levels for contaminant concentrations should be established.
b) A national data bank on contaminants should be established which will contribute to State of the Environment Reporting.
c) "Hot spots" (if any) should be identified.
d) Trends (if any) should be identified.

3. If successful, the monitoring network should be expanded to include more than shellfish growing areas, as well as address broader marine environmental considerations including health of the organisms.

QUESTIONS FOR MUSSEL WATCH EXPERT REVIEW

SAMPLING DESIGN

1. What are the criteria for "target locations"? Should sampling sites be sub-tidal?

2. What is the rationale behind number of samples (ie, number of sites, number of animals per site, number of replicates per site or study)? What is the minimum sample mass for analysis?

3. What is the rationale for the size and age of samples for collection? What is the appropriate size/age class?

4. Should sediments be collected in conjunction with shellfish? If so, how much sediment? Top 1 cm?

5. Have there been any statistical studies to determine the optimal numbers of organisms per sample or optimal sediment quantities? What are the strengths and weaknesses of pooling samples?

6. Is the use of caged mussels a good approach for monitoring bioaccumulation of toxic chemicals?

7. Is the use of transplanted mussels recommended?

8. How does the reproductive cycle of bivalve molluscs exert an influence on tissue concentrations of contaminants? What should the timing of sampling be to minimize this influence?

9. What other physiological measures should be considered to normalize data?

TARGET SPECIES

1. What target species should be sampled?

2. Will one species of bivalve mollusc bioaccumulate all contaminants of concern?

3. Can you compare data between species? e.g. oysters bioaccumulate zinc to much greater concentrations than bivalves.

4. Should mussels be the organism of choice if the main program objective is determining contaminant levels in shellfish growing waters which are used primarily for oyster and clam production?

CONTAMINANTS OF CONCERN

1. What contaminants should be measured? What priority? Do regional needs determine the priority?

2. Are there contaminant interactions? If so, do they affect bioaccumulation or its measurement?

3. Is the measurement of biological parameters (detectable life functions such as: respiration; valve movement; Body Condition Index; adenylate Energy Charge, and the general state of health of the population) recommended?

PROTOCOLS

1. What issues should be addressed in specifying a tissue preparation protocol?

2. What protocols are recommended for analysis of priority pollutants in tissue?

3. Does freezing the tissue at -20°C affect the recovery of contaminants? Are ultra-low (-60°C) temperature freezers recommended?

STATISTICAL TECHNIQUES

1. What statistical techniques are most useful in the analysis of the data for: trend analysis, sample variability, geographic variability, species variability?

QA/QC

1. What should a good QA/QC program include with respect to sampling, analytical and statistical considerations?

COSTING

1. What are the costs involved in a mussel watch program? (i.e. for sampling, analysis and reporting)

APPENDIX II: Participant's List

NAME/AFFILIATION	ADDRESS	PHONE/FAX NO.
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John Machell	Environment Canada 45 Alderney Drive Dartmouth, N.S. B2Y 2N6	(902) 426-4570
Jack Uthe	Marine Chemistry Division Dept. of Fisheries & Oceans P.O. Box 550 Halifax, N.S. B3J 2S7	(902) 426-6277
Bill Ernst	Environment Canada 45 Alderney Drive Dartmouth, N.S. B2Y 2N6	(902) 426-6141
Ron Pierce (replacing Jean Piuze)	Dept. of Fisheries & Oceans 200 Kent Street Ottawa, Ontario K1A 0E6	(613) 990-9480
Peter Wells	Environment Canada 45 Alderney Drive Dartmouth, N.S. B2Y 2N6	(902) 426-9632
Bruce Kay	Environment Canada 224 W. Esplanade N. Vancouver, B.C. V7M 3H7	(604) 666-2736

NAME/AFFILIATION	ADDRESS	PHONE/FAX NO.
Lee Harding	Environment Canada 224 W. Esplanade N. Vancouver, B.C. V7M 3H7	(604) 666-2917
Janice Smith	National Water Research Institute P.O. Box 5050, 867 Lakeshore Rd. Burlington, Ontario L7R 4A6	(416) 336-4685 (416) 336-4989 Fax
Wayne Barchard	Environment Canada 45 Alderney Drive Dartmouth, N.S. B2Y 2N6	(902) 426-8304 (902) 426-9709 Fax
Rudy Chiang	Dept. of Fisheries & Oceans Fish Inspection Branch 2250 S. Boundary Road Burnaby, B.C. V5M 4L9	(604) 666-3150 (604) 666-4440 Fax
Gunnar Lauenstein	NOAA N/OMA32 6001 Executive Blvd. Rm. 312 Rockville, Maryland 20851	(301) 443-8655
Paul Kluckner	Environment Canada C&P Laboratories 4195 Marine Drive West Vancouver, B.C. V7V 1N8	(604) 666-6767
John Karau	Environment Canada Place Vincent Massey Ottawa, Ontario K1A 0H3	(819) 953-1699

MUSSEL WATCH STEERING COMMITTEE TERMS OF REFERENCE

NOVEMBER 27, 1990

PROTOCOLS FOR PILOT SCALE MUSSEL WATCH MONITORING PROJECT FOR SHELLFISH GROWING AREAS

Background

In early 1990 the Shellfish Water Quality Protection Program of Environment Canada contracted to Seakem Oceanography Ltd. to prepare a review document of Mussel Watch Programs in North America and elsewhere in the world. The purpose of the contract was to provide an evaluation of the benefits and problems associated with Mussel Watch programs, and provide a basis for further discussions within Environment Canada and other federal agencies (such as the Department of Fisheries and Oceans) as to the potential for beginning a Mussel Watch program in Canada.

Using the draft report "National Mussel Watch Program. Phase I. Overview of Existing Mussel Watch Programs" as background material, a Panel Discussion was conducted in conjunction with the 17th Annual Aquatic Toxicity Workshop in Vancouver to evaluate specific objectives for a Mussel Watch program and discuss a number of technical issues raised by the Seakem report. The Panel was comprised of a number of experts in Mussel Watch studies as well as program managers for the Shellfish Program and other marine environmental quality programs. The conclusions and recommendations of this Panel were as follows:

1. The Panel concluded there was sufficient merit in proceeding with a program to measure contaminant levels in bivalve molluscs and sediments from shellfish growing areas and other reference sites.

2. It was agreed that a Steering Committee be struck to develop a protocol document for a Mussel Watch Pilot Project which would address: site selection; target species; sampling/collection requirements; contaminants of concern; analytical procedures; QA/QC; statistical requirements and constraints; physiological considerations and other measurements. Several of the Panel members agreed to participate in this Steering Committee. This protocol document would use the NOAA Status and Trends Program Mussel Watch as the reference program.

3. A Terms of Reference be developed for the Steering Committee within 2-3 weeks with a draft protocol document to be available by the end of January 1991 for review by Shellfish Program managers.

4. A final document detailing protocols, integration with other programs and initial and annual costs be completed by the end of March 1991.

Objective

Develop a protocol document which will serve as the basis for a pilot "Mussel Watch" project to measure baseline contaminant levels and assess trends in bivalve shellfish growing areas under the requirements of the Canadian Shellfish Sanitation Program.

Terms of Reference

1. The following documents will be considered the standard reference documents for the Pilot project (subject to revision):

Protocols:

Shigenaka, G. and G.G. Lauenstein. 1988. National Status and Trends Program for Marine Environmental Quality: Benthic Surveillance and Mussel Watch Projects Sampling Protocols. NOAA Technical Memorandum NOS OMA 40.

NOAA Office of Oceanography and Marine Assessment, Ocean Assessments Division, Rockville, MD. 12pp.

MacLeod, W. D., Jr., D. W. Brown, A. J. Friedman, D. G. Burrows, O. Maynes, R. W. Pearce, C. A. Wigren, and R. G. Bogar. 1985. Standard Analytical Procedures of the NOAA National Analytical Facility, 1985-1986: Extractable toxic organic compounds. 2nd edition. NOAA Technical Memorandum NMFS F/NWC-92, 121 pp.

Krahn, M. M., C. A. Wigren, R. W. Pearce, L. K. Moore, R. G. Bogar, MacLeod, W. D., Jr., S. Chan, and D. W. Brown. 1988. Standard Analytical Procedures of the NOAA National Analytical Facility, 1988: New HPLC Clean-up and Revised Extraction Procedures for Organic Contaminants. NOAA Technical Memorandum NMFS F/NWC-153, 52 pp.

Lauenstein, G. G., S. A. Wise, R. Zeisler, B. J. Koster, M. M. Schantz, and S. L. Golembiewska. 1987. National Status & Trends Program for Marine Environmental Quality. Specimen Bank Project: Field Manual. NOAA Technical Memorandum NOS OMA 37.

Battelle New England Marine Research Laboratory. 1986. Phase II Field Manual for Collection of Bivalve Molluscs and Surficial Sediments, and Performance of Analyses for Organic Chemicals and Toxic Trace Elements. Prepared for U.S. Department of Commerce National Oceanic and Atmospheric Administration Status and Trends Mussel Watch Program. Contract No. 50-DGNC-5-0263.

Reporting:

NOAA, 1989. National Status & Trends Program for Marine Environmental Quality. Progress Report. A Summary of Data on Tissue Contamination from the First Three Years (1986-1988) of the Mussel Watch Project. NOAA Technical Memorandum NOS OMA 49.

NOAA, 1988. National Status & Trends Program for Marine Environmental Quality. Progress Report. A Summary of Selected Data on Chemical Contaminants in Sediments Collected During 1984, 1985, 1986, and 1987. NOAA Technical Memorandum NOS OMA 44.

NOAA, 1989. National Status & Trends Program for Marine Environmental Quality. Progress Report. A Summary of Data on Individual Organic Contaminants in Sediments Collected During 1984, 1985, 1986, and 1987. NOAA Technical Memorandum NOS OMA 47.

Lauenstein, G. G., A. Robertson, and T. P. O'Connor. 1990. Comparison of Trace Metal Data in Mussels and Oysters from a Mussel Watch Programme of the 1970s with those from a 1980s Programme. Marine Pollution Bulletin, Volume 21, No.9, pp 440-447.

Using these documents, identify
(a) the critical components,
(b) deficiencies and errors,
(c) alternate/improved protocols and,
(d) additional considerations or requirements for the following subject areas:

i) Site Selection
ii) Target species
iii) Sampling/collection requirements (including sediments)
iv) Contaminants of concern
v) Analytical procedures
vi) QA/QC
vii) Statistical requirements and constraints (both for analytical results and data treatment)
viii) Other measurements (eg. scope for growth, physiological considerations).

Project Schedule

1. Steering Committee conference call in early to mid-December.

2. Provide comments to the Project Coordinator for compilation and distribution back to the Steering Committee by January 31, 1991.

3. Provide final review comments to Project Coordinator by February 28, 1991.

4. Draft protocol document circulated to Steering Committee and Shellfish Program Managers for review by March 31, 1991.

5. Final protocol document prepared by April 15, 1991.

Steering Committee Members

Dr. Glenn Atkinson, Environment Canada
Dr. Lloyd Dickey, Bedford Institute, Department of Fisheries and Oceans
Mr. Ken Freeman, Bedford Institute, Department of Fisheries and Oceans
Mr. Paul Kluckner, Environment Canada
Dr. Gunnar Lauenstein, NOAA, National Status and Trends Program
Dr. Paul Lobel, Memorial University, Newfoundland
Dr. Jack Uthe, Department of Fisheries and Oceans

Project Coordinator: Bruce Kay, Environment Canada

APPENDIX 3

MARINE ENVIRONMENTAL QUALITY ADVISORY GROUP

MINUTES OF MEETING, 8 November 1990, Vancouver, B.C.

Chair: Lee Harding, Environment Canada, Conservation and Protection, Vancouver

Members attending:

Wayne Barchard, Environment Canada, Conservation and Protection, MEB, Dartmouth
Terry McRae, Environment Canada, SOE Reporting, Ottawa
Ed Porter, Environment Canada, Environmental Protection, Yellowknife
Peter Wells, Environment Canada, Conservation and Protection, Dartmouth

Other participants:

Richard Addison, Fisheries and Oceans Canada, BIO, Dartmouth
Glen Atkinson, Environment Canada, Ottawa
Robert Bisson, Environment Canada, National Water Resources Institute, Burlington
Ron Buchanan, B.C. Ministry of Environment, Victoria
Phil Cohen, Environment Canada, Office of the Science Advisor, Ottawa
John Elliott, Environment Canada, Canadian Wildlife Service, Delta
Darcy Goyette, Environment Canada, North Vancouver
John Karau, Environment Canada, Marine Environment Division, Ottawa
Bruce Kay, Environment Canada, North Vancouver
Paul Laramée, Environment Canada, Environment Protection, Montreal
Colin Levings, Fisheries and Oceans Canada, West Vancouver
Karen Lloyd, Environment Canada, Canadian Wildlife Service, National Water Resources Institute, Hull
John Machell, Environment Canada, Dartmouth
Theresa Morton, B.C. Health Protection Branch, Burnaby
John F. Paul, U.S. Environment Protection Agency, Narragansett, RI
Ron Pierce, Fisheries and Oceans Canada, Ottawa
Martin Pomeroy, Environment Canada, North Vancouver
Linda Porebski, Environment Canada, Marine Environment Division, Ottawa
Jerry Payne, Fisheries and Oceans Canada, St John's
Les Swain, B.C. Ministry of the Environment, Victoria
Jack Uthe, Fisheries and Oceans Canada, Halifax

Recorder: Richard Banner, Polestar Communications

The chair called the meeting to order at approximately 0900 with a review of the agenda and a note that Agenda Item 2, Action items from last meeting, would not be discussed as there had been no previous meeting.

BACKGROUND

The chair reviewed the history of the Marine Environmental Quality Monitoring program, noting as well that the history was included in the information package that had been sent out to meeting participants. In his review, he pointed out that the MEQ monitoring program had two principal objectives:

- To supply government departments with the ongoing monitoring information they require in order to evaluate the effectiveness of their marine programs including in particular the Marine Environmental Quality Management Framework.
- To supply information in keeping with a Cabinet statement that Canadians have a right to know the quality of the marine environment.

The chair noted other monitoring programs that gather information for specific purposes, such as broad-scale shellfish and seabird monitoring, numerous site- or project-specific assessments that sometimes extend over some time, and pulp mill effluent monitoring. He said that few meet a rigorous definition of environmental quality monitoring, i.e., an *a priori* design for collection of information at prearranged times at regular and ongoing intervals. Several participants said that a great deal of specific information is available from a variety of sources, and Bruce Kay stated that by using these sources it was possible to assess the state of the environment, but not changes over time. It was agreed that the shellfish and contaminants in seabird egg monitoring programs were useful for long term monitoring because they do follow rigorous protocols to test hypotheses.

The chair concluded that there is still no simple way to gather and assimilate information on marine environmental quality in a way that meets the two above objectives.

Peter Wells reviewed the UNEP "Montreal Guidelines" on the protection of the marine environment from pollution from land-based sources, signed by Canada in 1985. (The Guidelines were included in material mailed to MEQAG participants.) Wells said the Guidelines provided the organizational framework around which the marine environmental protection strategy was based. The guidelines indicate the establishment of an internationally complementary program for monitoring and the storage and exchange of data based on compatible procedures and methods. Monitoring should include collection of data on national conditions; identification of inputs into the marine environment; identification of levels of pollutants and their fates and effects; and an evaluation of the effectiveness of control measures relative to environmental objectives. Annex III to the Guidelines discusses the definition and objectives of monitoring: to monitor the resources to be protected; to gather information on inputs; to establish baselines; to observe trends by ongoing sampling and analysis.

Jerry Payne added that it is also important in monitoring to judge the economic value of the resource being protected and the potential impact on human health in order to avoid viewing everything as a resource to be protected. Lee Harding and others added that non-economic values are also significant, such as the "contingent" value of losing a resource, and amenity values such as recreation, shipping and even sewage dumping.

The scope of a proposed national MEQ monitoring program was discussed, and it was agreed that as a national program it should focus on issues or indices of environmental quality that have national relevance; purely regional issues, or site/project-specific assessments, should not have to be modified to meet some national criteria, nor be added to a national MEQ database.

The chair summarized by noting the following are accepted:

1. The contaminants in seabirds program and shellfish monitoring program should be maintained, and possibly enhanced.
2. Existing monitoring programs should not be duplicated, but a data exchange system may be necessary to ensure access to data.
3. As a national program, MEQ Monitoring will not recommend involvement in purely regional monitoring.

Richard Banner reported on the marine environmental monitoring workshop presented at the Aquatic Toxicity Workshop on 7 November 1990. Lee Harding added that the three groups principally involved in environmental monitoring interact with each other. Management tells scientists what it wants, and scientists tell management what they need. Scientists give information to the public, and the public uses it to tell management what it wants. The interaction of these different forces defines what is needed in a monitoring program. Lee Harding also pointed out that agreements involving more than one jurisdiction help preserve continuity by creating obligations that extend beyond immediate concerns. Peter Wells added that at a meeting in Victoria in November 1989, senior managers drafted a federal framework on marine environmental quality. This framework included conducting marine research, monitoring and assessment studies; establishing and maintaining monitoring programs; and providing information on the state of the environment. John Karau pointed out that even with the existing networking and commitments, large gaps exist in the information, and those gaps need to be identified. He added that access to information remains an issue for the public. Ron Buchanan noted that a major problem remains in data management to improve access to information which is available.

The chair observed that three different information-based programs operate and interact concurrently: monitoring, the guideline development process, and state of the environment reporting. He said the objective of the meeting is not to work out the details of all of these, but to outline the major elements of a monitoring program for presentation to senior management.

PURPOSE OF MONITORING

The chair directed the meeting to the second point of the "Questions for MEQAG Monitoring Task Force Meeting", the purpose of monitoring (Annex 1). Wayne Barchard said that the objectives of the public are not necessarily recognized in the existing processes. John Karau said that public values are incorporated in setting environmental guidelines and objectives through public consultations, polls and information volunteered by the public.

John Karau said that trends in habitat and loss of habitat should be considered. Jerry Payne said that the split between DFO objectives and DOE objectives seemed unnecessary. Les Swain added that the objectives appear to have a long-term focus, and that existing monitoring programs such as pulp mill monitoring should also be recognized.

Colin Levings said that the use of the word "health" in the proposed definition of purpose was vague, and Peter Wells suggested the following MEQ definition, from the paper "Control of Marine Pollution from Land Based Sources" was becoming widely accepted by scientists:

> Marine environmental quality can be described as the condition of a marine environment measured relative to the intended use of that environment. It is usually assessed quantitatively on temporal and spacial scales and it is measured relative to objectives and limits set by environmental, health and resource agencies. Environmental quality may include subjective perceptions of the degree of acceptability. Its measurement requires having sensitive and interpretable indices of condition and change.

It was generally accepted that this definition of environmental quality relative to defined standards might be adequate for scientists, but that managers and the public prefer a more easily comprehensible definition, such as "health" or "state" of the oceans, vague though these may be.

Richard Addison added that from a research approach, a reference to indices was useful, and that "scope for growth" is one generally applicable measure of marine health. Jack Uthe said that no single measure could be used, and that it was generally accepted that a battery of tests and measures would be needed. The chair suggested that the task force could propose a set of tests, including scope for growth. Jerry Payne said that the basic tests needed are generally known and used in programs such as NOAA Status and Trends and the North Sea Master Monitoring Program. He said that three basic concepts could be applied: EROD, sediment chemistry and sediment bioassaying. Peter Wells said that the question was whether to test for known problems only or to use more general tests to discover environmental problems and the select which specific problems to look for. Phil Cohen queried whether the purpose of a monitoring problem was to tell the public the state of the environment or to watch for threats that could be coming in the future. John Elliott questioned whether the focus should be a global monitoring program or a series of more specifically defined programs such as Arctic monitoring, Strait of Georgia monitoring, etc.

The chair summarized the discussion as follows:

1. The purpose of monitoring includes
   - evaluating controls and the need for controls;
   - identifying trends and;
   - protecting the environment against threats.

2. It was agreed that evaluation of ecosystem health should be part of a monitoring program.

3. Tools are available to monitor ecosystem health.

4. A technical sub-committee should be struck to prepare a proposal for management defining the specific tests needed.

MONITORING INDICATORS

Terry McRae discussed the use of indicators in State of Environment monitoring. He said that State of the Environment data includes both ambient environmental conditions and site specific data; it is not a separate category as the report on the monitoring workshop suggests. He said that the use of reference thresholds is essential in defining targets and objectives, and that to evaluate the state of the environment, indicators are needed to show how well those objectives are being met.

Bruce Kay discussed the mussel watch and shellfish monitoring programs as indicators of the state of the environment. He said one of the objectives of the Mussel Watch program is to check whether certain standards of environmental quality are being met in shellfish harvesting areas. This program could be expanded and better integrated by establishing appropriate protocols on site selection, species, sampling, etc. He said that a drafting committee to develop protocols has been established and should introduce a draft by January. He said, however, that the sampling protocols for monitoring contaminants in commercial shellfish areas may not completely satisfy a need for ambient MEQ monitoring; however, reference stations could be selected which satisfy basic needs. The program should be integrated with other environmental data, such as sediment studies, seabird monitoring, etc.

The chair summarized by saying that

1. Work should be undertaken to link together the various program elements into a more comprehensive monitoring system.
2. The Monitoring Task Force should continue to communicate with the Guidelines Task Force.

PROGRAM ELEMENTS

The chair asked the meeting to review the Program Elements listed in Questions for MEQAG Monitoring Task Force Meeting (Annex 1). He said that the elements came from an Environment Canada inventory prepared in 1988 for the MEQ Working Group and from comments on the inventory by the Fisheries Science Branch. (As discussion proceeded from subheading to subheading, this report assumes that points not raised in discussion were accepted by the participants.)

Sources of Pollution

The chair noted that there is no good understanding of non-point sources of pollution, such as urban runoff and storm sewers, although some study of these sources is being done in Atlantic Canada. It was pointed out that riverine sources are also not well known, except on major rivers, although pollution is not believed to be a serious problem except on major rivers or those with significant pollution sources. This raises a question of scale and the resolution to which smaller rivers can be monitored. Richard Addison added that as the major sources of organic pollution are often atmospheric, a detailed study of riverine inputs would not be useful in those cases.

The chair concluded that

1. Riverine and atmospheric input data (or rather, interpreted results) from the responsible agencies should be included in an MEQ monitoring network. Interagency agreements may be useful to ensure regular transmission of such data.
2. If existing atmospheric and riverine data are found not to be adequate to describe pollution sources to the marine environment, enhanced monitoring may be negotiated between agencies.

Sinks of Pollutants

The chair observed that several Canadian programs already monitor contaminants in biota, sediment and water. Wayne Barchard said that most programs do not provide useful background base level data; but could if they were formalized with agreements to continue on a regular basis, and if standard sampling and analytical protocols were agreed upon to ensure compatibility of data between regions and among the different groups doing the work. John Karau suggested control sites could be built into the Ocean Dumping Program, and that the aggregate of all control sites for ocean dumping, mussel watch and EEM programs will provide ambient MEQ data.

Jerry Payne said that offshore sites are not monitored as much as coastal sites, although they can be economically valuable sites. It was suggested that ships passing through the open ocean on other duties could take water and sediment samples as well as samples to monitor the state of benthic communities and pollutants. The use of marine mammals to monitor the offshore environment was considered. John Elliott said that seabirds alone would not provide enough useful information as no background levels have been established and the large number of variables would make information difficult to interpret. Peter Wells said that CWS believes that seabird studies can usefully augment contamination and environmental studies. It was said that with statistical analysis, seabird studies could show trends and changes in the environment. Jack Uthe said some aspects of marine ecosystem health are not amenable to routine monitoring and derivation of meaningful indices, at least with the current state of knowledge; and that research on status of marine ecosystem health should be closely linked with more routine monitoring and could be used as a basis for program design.

The chair summarized saying

1. Existing monitoring programs should be formalized with standardized sampling and analytical protocols to the extent feasible and appropriate, by regional committees. These could be linked nationally to ensure compatibility of data between regions.
2. The Ocean Dumping Control Program should establish control sites in conjunction with environmental effects monitoring and the Mussel Watch program.
3. Offshore sites should be added into existing monitoring programs including both chemical and biological studies.

4. The Task Force should support CWS studies on seabirds to monitor the marine environment, and should establish links to exchange information.
5. The scale of the studies needed is still unresolved.
6. Research on fate and effects of pollutants on marine ecosystems should be encouraged, and will be an important component of periodic marine ecosystem health assessments.

The meeting broke for lunch at approximately 1200 h and reconvened at approximately 1330 h.

Effects of pollutants

Richard Addison said the program objectives raise questions of whether the program should monitor distribution of chemicals over time and space or effects of chemicals in the marine environment; the scale and resolution of monitoring studies; and the time periods over which studies should be carried out. John Karau said that transport, fate and effects of chemicals should be studied, as well as habitat and habitat destruction, but that more expert advice is needed to decide questions of scale. Jerry Payne said that those items are needed as well as more regional items such as resource levels. Wayne Barchard said that the question is whether the program is to monitor point sources for the purpose of control or to monitor ecosystem health. John Karau said the Environment Canada is interested mainly in broad trends for SOE reporting. Lee Harding noted that evaluation of marine program effectiveness is also an objective. Peter Wells said both are valid objectives but that trying to blend the two will lead to problems in choosing an appropriate scale; he said the program should monitor on a national scale what the transport, fate and effects of pollutants are.

The chair summarized saying that the broad principles and objectives of the program include both ecosystem health and effects of controls. The specific questions of sites, frequency and scale will depend on the environmental variables and the threats to the environment that the program focuses on. The question that remains is what goes into the program to meet the objectives. Existing programs already show data on compliance and management but not environmental response.

John Karau said that data was still needed on effectiveness of controls. Richard Addison said that two separate strategies were needed, one to monitor broad changes in the environment, and one to monitor separate contaminants. Jerry Payne said that management has defined reporting on the state of environment as part of the job of environment scientists. Peter Wells said that it is difficult to make general statements about environmental quality when the data only support narrow localized conclusions. Richard Addison said that different objectives might be met in the same program. Lee Harding noted that progress has been made in that consistent seabird and shellfish monitoring programs are in place, and monitoring of pulp mill receiving waters is being put in place. Jerry Payne said that although these might not provide detailed site-specific data, they do provide a general picture. John Elliott said that seabird studies can point to general issues at a very low cost, and that when broad issues are identified, more detailed studies can be carried out. Phil Cohen said that there was no consensus on monitoring objectives and that the group should make use of whatever imperfect statements exist but Wayne Barchard objected that this approach could lead to fallacious conclusions. The chair closed the discussion without a complete consensus being reached; however, there was general agreement that the program should monitor both ecosystem health and effectiveness of controls.

Effects of coastal restructuring

This has been an important issue for the Atlantic Region, and in specific Pacific and Yukon Region estuaries, such as the Fraser River Estuary. The main concerns are loss of fish and wildlife habitat and physical changes that alter currents and salinity. John Karau said that it is important to monitor habitat and habitat change.

Effects of global change

"External" influences on marine environmental quality, i.e., those that do not result from local or regional human activities, were discussed. They included sea level rise, sea temperature change (that might occur from global warming or phenomena such as the warm El Niño current that periodically wreaks havoc in north Pacific ocean) that can have dramatic effects on phytoplankton and other marine life, and altered runoff patterns that change estuarine circulation. It was agreed that marine environmental quality can not be assessed without knowledge of these external factors. Peter Wells said that existing programs should be expanded to include monitoring of global change.

Management responses

Several speakers commented that management data such as compliance records or closures and openings are not reliable indicators as they may reflect effort as much as environmental change. However, effectiveness of management might be evaluated by monitoring variances, closures, permits and so forth. Discussion on this subject was inconclusive.

DESIGN CONSIDERATIONS

Peter Wells said that reports on monitoring programs do not make it clear how QA/QC standards are being met and that a protocol for standardized procedures is needed. Jerry Payne pointed out that methods are evolving over time, and that consistency over time is not possible. Peter Wells said that what is needed is an assurance that similar tests will produce similar results. Ron Buchanan said that quality assurance is known to be a problem with existing programs in Canada.

The chair summarized that

1. QA/QC procedures are needed on a nationally standardized basis.
2. Statistical analysis should be part of the program design.

It was noted, however, that biological identification and taxonomy could be difficult to assure with current budgetary restrictions. Also, a need for reference samples of constituent chemicals continues.

DATA MANAGEMENT

The chair suggested that new funding may be needed for data management, as there is no general consistent data base at present. Richard Addison noted that DFO has the Marine Environmental Data System in which environmental data is recorded. It might be used as a model or a beginning for a data base on environmental quality. Wayne Barchard said that a focus is needed on consistent data management rather than creating a new data base. Peter Wells suggested that managers of the various data bases should meet to compare procedures and standards to ensure that information is accessible to various users. John Elliott described a Swedish national data system that is very easy to access and contains a large amount of information. Lee Harding suggested that access through electronic bulletin boards might be feasible. It was also suggested that data be archived on optical disk at regular intervals, as water quality data is now in the Atlantic region.

PROCESS QUESTIONS

John Karau discussed the question of a mechanism for coordinating the Monitoring Program. He suggested that the Inter-departmental Committee on Oceans (ICO) has an interest in ocean monitoring, appropriate expertise, a committee structure that would be appropriate and in addition works on a level that is sufficiently senior to ensure full collaboration and co-operation. Wayne Barchard queried who would have responsibility to integrate data and produce reports under the monitoring program. The chair said that if no existing agency has the resources and the time, it may be necessary to issue a contract for the work.

He added that since the monitoring program would have to link with existing programs, management and continuity would be needed, and might be protected by formal agreements between participating agencies.

Wayne Barchard suggested that a visual model of the monitoring program be prepared for use as a tool to explain how the different components of the program work together. The chair said that he will prepare a graphic chart which would function as a visual model.

SUMMARY

The chair agreed to draft a set of objectives consistent with the two broad purposes of the Task Force, and to circulate the draft for review. A draft of a monitoring program design will also be prepared and circulated for discussion at the next meeting.

The meeting concluded at 1500 h.

ANNEX 1.

QUESTIONS FOR MEQAG MONITORING TASKFORCE MEETING

DEFINITION

Monitoring marine environmental quality is the collection, analysis and interpretation, at regular intervals, of data to determine in what areas (environmental compartments, ecosystem components) the quality of the marine environment is improving or declining.

PURPOSE

1. Why monitor?
   DOE:
   - -to evaluate the effectiveness of controls
   - -to identify the need for new controls
   - -to determine the trends in contaminant levels
   - -to evaluate the "health" of marine ecosystems
   - -to advise Canadians on the state of the marine environment.

   DFO:
   - -to determine the health/vigour of fish stocks
   - -to identify causes of declines in fish stocks
   - -to evaluate global changes (climate, ocean currents, ocean chemistry) affecting, or that may affect, Canada's oceans
   - -to evaluate the "health" of marine ecosystems

PRODUCTS

1. What do we want out of it?
   - -SOE reports (5 year; annual; regional/national; issue-specific); Canadians know and understand the quality of the marine environment
   - -improved scientific basis for pollution control decision-making (be specific: what problem do we now have that would be solved by the program?)
   - -improved scientific basis coastal development decision-making
   - -feedback: decision-makers know the environmental implications of their decisions.
   - -independent audit of pollution regulation programs

PROCESS

1. How should an MEQ Monitoring program be developed? Some options (not mutually exclusive):
   - -As a working group under ICO?
   - -As a working group under MEQAG?
   - -By regional staff of each agency, with programs linked by data exchange agreements?
   - -via CCME (similar to MEQ Guidelines process)

2. Can an MEQ program be developed before national marome SOE Indicators are established?

3. How should an MEQ Monitoring program be funded?
   - -Green Plan
   - -A Base
   - -TB Submission
   - -Partnerships with universities, industry
   - -Polluter pays (e.g., EEM programs to be enforceable under Pulp and Paper Effluent Regulations)
   - -by Provinces/Territories
   - -by agencies with regulatory authority (e.g., Provinces/Territories for most land-based discharges; DFO for discharges permitted under the Fisheries Act; CCG for ship discharges; DOE for ocean dumping under CEPA Part VI; COGLA for offshore drilling; etc.)

4. How can programs of various agencies be coordinated (Great Lakes, St. Lawrence, BIMP, FREMP, Gulf of Maine, Puget Sound, ICES)?

5. Is there a need for other Regional Seas programs like the Gulf of Maine program (e.g., Gulf of Georgia, Beaufort Sea, Davis Strait)?

6. Do we need a new MEQ monitoring program (or components thereof) or should we focus on managing and integrating the data from the program we now have?

ELEMENTS

Sources of pollution

- -Volume/characteristics of land-based discharges
- -volume/characteristics of ocean dumped materials
- -number/type/volume of spills
- -riverine inputs
- -atmospheric inputs
- -etc.

Sinks of pollutants

- -Biota (what species? what trophic levels? what resources?)-integrate contaminants over short time
    - -Fish
    - -seabirds & shorebirds
    - -marine mammals
    - -shellfish (commercial species in commercial harvest areas or "mussel watch")
- -Sediment: integrate contaminants over longer time; ok for hydrophobic contaminants
- -Water - medium of respiration for most marine organisms; ok for hydrophillic contaminants

Effects of pollutants

- -ecosystem health indicators (indicator species; primary productivity)
- -incidence/severity/extent of hypoxia
- -diagnostic symptoms (imposex in gastropods; liver lesions in sole; shell thickening in oysters; shell thinning/chick edema in birds; MFO/EROD induction in fish)
- -ecosystem health indices (diversity, abundance, community structure & function)
- -toxicity (suite of tests)

Effects of coastal restructuring

- -turbidity due to discharges particulate matter
- -temperature and salinity changes due to engineering works
- -decreased circulation in estuaries due to sedimentation
- -altered sedimentation/scour (erosion) patterns due to coastal developments
- -changes in fish habitat quantity/quality
- -changes in wildlife habitat quantity/quality

Effects of global change

- -changes in sea levels
- -changes in assimilative capacity
- -changes in sea surface temperatures, and consequent changes in biological communities (proliferation of nuisance species; introduction of ecologically or economically harmful, exotic species)
- -changes in resource distribution/abundance

Management Responses

- -Closure/re-opening of fisheries
- -Rejection of ocean dumping application

DESIGN CONSIDERATIONS

1. Should QA/QC be managed by data contributors, or by a central group?

2. To what level of detail should parameters (variables), frequency, number of replicates etc. be specified in a national program? Do we need national protocols for sample collection, biological identifications, procedures to reduce variation from sex, age and reproductive status of specimens, etc.?

3. Should statistical analysis be a part of the *a priori* program design (and should it be a part of the data management component)?

DATA MANAGEMENT

1. A national database, or regional, nationally consistent databases?

2. Is there a need for protocols on database structure or input/output data format?

3. How can QA/QC results be tracked with environmental data?

4. What controls are needed on access to the data?